STRUCTURE AND FUNCTIONING OF TRIBUTARY WATERSHED ECOSYSTEMS IN THE EASTERN RIVERS AND MOUNTAINS NETWORK: CONCEPTUAL MODELS AND VITAL SIGNS MONITORING

Robert P. Brooks[1], Craig Synder[2], and Mark M. Brinson[3]
([1]with assistance from Wendy Mahaney, Jennifer Rubbo, and Suzy Laubscher)

[1]Penn State Cooperative Wetlands Center, Department of Geography, 302 Walker Building, University Park, PA 16802
[2]U.S. Geological Survey, Leetown Science Center, Kearneysville, WV 25430
[3]Department of Biology, East Carolina University, Greenville, NC 27858

June 2005

U.S. Department of the Interior
National Park Service
Eastern Rivers and Mountains Network
State College, PA

Table of Contents

INTRODUCTION AND BACKGROUND

Purpose and Content of this Report

The importance of the tributary portion of watersheds (i.e., tributary watersheds) to the overall health of aquatic ecosystems cannot be over emphasized. In the eastern U.S., tributary watersheds typically comprise about 67-75% of the contributing area of any given watershed (Table 1). That is, the combined areas of terrestrial habitats, wetlands, floodplains, and headwater streams occupy two-thirds to three-quarters of the total drainage basin for larger rivers. Given this influence on downstream portions of large river watersheds, understanding the impacts of human activities on the ecological structure and function of tributary watersheds is foundational for optimizing their conservation and management.

Table 1. Example of proportion of headwaters in a watershed in the eastern U.S.

Strahler Order	Number of Random Locations	Percent of Total Stream Length	Mean Watershed Area (ha)
1	31	62	287
2	9	18	796
3	3	6	2,524
4	3	6	10,790
5	1	2	44,354
6+	3	6	N/A
Total	50	100	

The conceptual model presented here, in narrative and graphical forms, represents an attempt to portray the diversity and complexity present in tributary watersheds. In the context of the National Park Service's Inventory and Monitoring (I&M) Program, these conceptual models seek to "promote communication and integration among scientists and managers from different disciplines during the vital signs selection process". Also, as envisioned for the Eastern Rivers and Mountains Network (ERMN; Table 2), this conceptual model is designed to emphasize the role of stressors in the alteration and degradation of these ecosystems. By summarizing our ecological understanding of these critical aquatic resources, and documenting the impacts of a range of anthropogenic stressors of these systems, an appropriate and useful set of indicators can be identified to portray the ecological integrity or condition of these ecosystems.

Table 2. Parks in the Eastern Rivers and Mountains Network (ERMN), with abbreviations.

ALPO	Allegheny Portage Railroad National Historical Site
BLUE	Bluestone National Scenic River
DEWA	Delaware Water Gap National Recreational Area
FONE	Fort Necessity National Battlefield
FRHI	Friendship Hill National Historical Site
GARI	Gauley River National Recreational Area
JOFL	Johnstown Flood National Memorial
NERI	New River Gorge National River
UPDE	Upper Delaware Scenic & Recreational River

Vital Signs Definition

Park vital signs are selected physical, chemical, and biological elements and process of park ecosystems that represent the overall health or condition of the park, known or hypothesized effects of stressors, or elements that have important human values. The elements and processes that are monitored are a subset of the total suite of natural resources that park managers are directed to preserve "unimpaired for future generations," including water, air, geological resources, plants and animals, and the various ecological, biological, and physical processes that act on those resources. Vital signs may occur at any level of organization including landscape, community, population, organism, or genetic level, and may be compositional (referring to the variety of elements in the system), structural (referring to the organization or pattern of the system), or functional (referring to ecological processes). For definitions, see National Park Service, http://science.nature.nps.gov/im/monitor/vsm.htm.

Goals of the I&M Vital Signs monitoring program:

- Determine status and trends in selected indicators of the condition of park ecosystems to allow managers to make better-informed decisions and to work more effectively with other agencies for the benefit of park resources.
- Provide early warning of abnormal conditions of selected resources to help develop effective mitigation measures and reduce costs of management.
- Provide data to better understand the dynamic nature and condition of park ecosystems and to provide reference points for comparisons with other altered environments.
- Provide data to meet certain legal and Congressional mandates related to natural resource protection and visitor enjoyment.
- Provide a means of measuring progress towards performance goals.

Ecological Scope

For this purpose, tributary watersheds are defined as a stream network consisting primarily of first and second order streams (at a 1:24,000 scale, Strahler 1952), and including where appropriate “zero” order streams which represent intermittent streams and ephemeral channels, and third order and occasionally fourth order streams depending on relative discharge. That is, we consider all aquatic resources from the upstream portion of the watershed (including uplands and wetlands) to mid-reach streams as part of the *tributary watershed*. We also recognize how tributary watersheds contribute significantly to the ecological integrity of larger rivers. It is essential to move away from considering streams in isolation from their surroundings, and integrate all components of aquatic ecosystems, including the associated wetlands, floodplains, riparian corridors, and the influence of contributing terrestrial areas. This is critical to understanding and protecting tributary watersheds because these headwater portions of larger watersheds are often subjected to a wide range of stressors.

In the ERMN, the component parks that contain significant amounts of tributary watershed resources are DEWA and NERI. Both have large numbers of low-order, headwater streams and upland wetlands. Headwater streams in these parks are largely situated within forested watersheds of relatively steep topography, have mostly constrained channels with little or no floodplains, and are underlain by a mosaic of surface geologies that fosters significant instream habitat diversity. All of the remaining parks within the ERMN (Table 2) have smaller amounts of tributary resources while the UPDE consists of an entirely riverine ecosystem. In addition, there is a suite of anthropogenic stresses common to the component parks within ERMN which include urbanization, agriculture, acidification from acid mine drainage and acid precipitation, exotic species, and forest pests.

What follows is a narrative and set of graphical models designed to illustrate the important natural processes that drive tributary watersheds, to describe the ecological contributions for tributaries to larger watersheds, and to indicate how the most anthropogenic stresses in this region are likely to alter tributary watersheds and ultimately degrade biological integrity. Finally, we hope to convey a list of potential measures that could be instituted in a long-term ecological monitoring program, termed “vital signs” by the National Park Service.

KEY ECOLOGICAL CONCEPTS ABOUT TRIBUTARY WATERSHEDS: A CONCEPTUAL MODEL

Background

There are many conceptual models of riverine systems in the literature, variously describing the physical, chemical, and biological components (see Vannote et al. 1980, Minshall et al. 1985, Ward 1989, Forman 1995, Ward and Tockner 2001, Thoms and Parsons 2002, Thorp et al. (in press)). It is not the intent of this document to comprehensively review these works and the plethora of papers that support and challenge these concepts, but rather to consider these concepts in light of the characteristics of tributary watersheds in ERMN, and how they relate to monitoring the condition of tributary watersheds and the impact of stressors upon them.

Historically, rivers have been variously portrayed as continuous linear (upstream-downstream) gradients (e.g, the River Continuum Concept, Vannote et al. 1980, Minshall et al. 1985), or as a series of distinct, interconnected habitat patches (e.g., Link Discontinuity Concept, Rice et al. 2001). In addition, alternative conceptual models have evolved seeking to classify stream networks (Frissell et al. 1986, Rosgen 1994), explain the physical heterogeneity of rivers (natural flow regimes, Poff et al. 1997; river discontinua, Poole 2002; network dynamics hypothesis, Benda et al. 2004), describe material cycling (riverine productivity model, Thorp and DeLong 1994, 2002; process domains, Montgomery 1999), and perturbations (intermediate disturbance hypothesis applied to rivers, Townsend et al. 1997). Of particular value to this discussion, are the ideas that characterize riverine ecosystems as a series of interconnected hydrogeomorphic patches (Church 2002, Poole 2002, Thoms and Parsons 2002, Thorp et al. (in press)) and the relationship of these dynamic patches to aquatic biodiversity (Wu and Loucks 1995, Townsend et al. 1997, Lake 2000, Ward and Tockner 2001, Thorp et al. (in press)).

Increasingly, these syntheses have begun to move beyond the stream or river channel alone, to incorporating linkages between streams and the landscape in which they flow, thus recognizing longitudinal, lateral, and vertical aspects of the tributary network (e.g., Forman 1995, Ward et al. 2002, Wiens 2002). Still missing, however, are attempts to create conceptual models that directly integrate stream, wetland, riparian, and terrestrial components for headwater or tributary watersheds. Within biogeographical constraints, species composition and biological integrity of tributary watersheds are the result of interactions among numerous important instream variables including flow regime, energy source, water quality, instream habitat, and biological interactions. Yet, these variables themselves are largely driven by processes that occur outside of the individual stream channel including weather and climate, geomorphology of the watershed (geology and terrain), and the structure and topology of the surrounding landscape. As we move downstream, instream characteristics are also determined by characteristics and processes in that occur in upstream areas. It is the magnitude and interplay between these vertical, lateral, and longitudinal processes that form the basis for most conceptual models of stream and riverine ecosystems.

For the purposes of this paper, the interactive relationships among the stream, wetland, riparian and upland components of watersheds for different stream orders are illustrated in Figures 1a- b, and 2a-c. A key feature of these illustrations is the relative contribution to the functioning of these systems by upstream portions of the watershed versus immediately adjacent or lateral components.

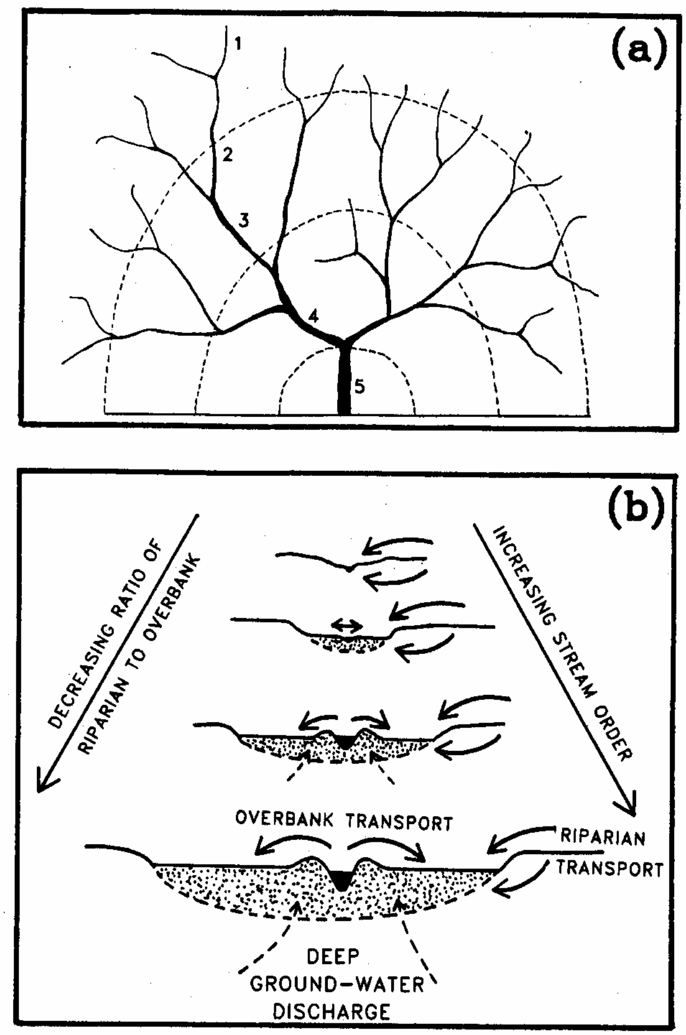
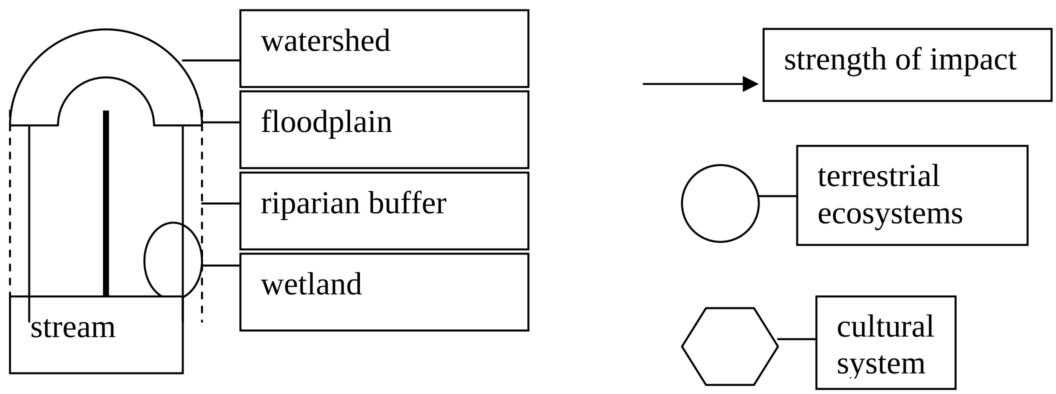


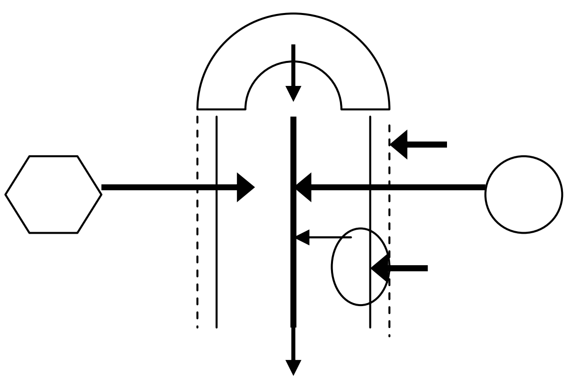
Figure 1a – 1b. . Representations of relative contributions of stream order to watershed area, flooding, and discharge.

Figures 2a-c. Model of elements in tributary watersheds and the relative impact of terrestrial and cultural systems as stream order increases.

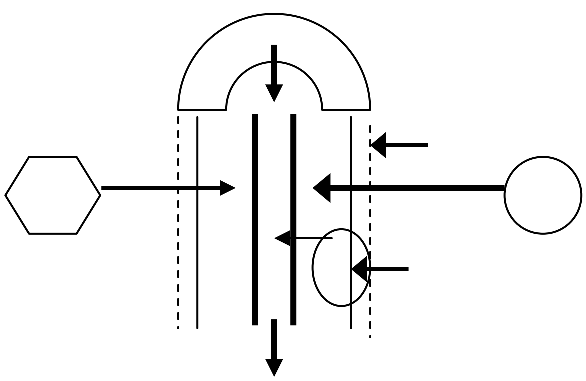
2a. Legend



2b. Headwater stream (2nd order with floodplain)



2c. Stream (3rd/4th order with floodplain)



General Conceptual Model

We begin with and modify Karr's (1991, 1999) basic conceptual model of stream ecosystems (Figure 3). The model focuses on biological and ecological endpoints ("integrity" applies to the condition of places/systems at one end of a continuum of human influence: those that support a biota that is the product of evolutionary and biogeographic processes with minimal influence from modern human society *sensu* Karr 1999) and five factors (Flow Regime, Water Quality, Energy Source, Biological Interactions, and Physical Habitat) that influence or modify the components of ecological integrity (Figure 3). These five factors provide a critical conceptual and analytical framework to judge the interactions of human activities and ecological change. We then show (Figure 4) how human activities (e.g., Agriculture, Recreation, Landuse, etc.) operate through a series of stressors (e.g., Altered Delivery of Water, Altered Water Quality, Increased Nutrients, etc.) to alter biogeochemical processes which influence one or more of the "five factors" thereby altering one or more element of "ecological integrity" (Figure 4).

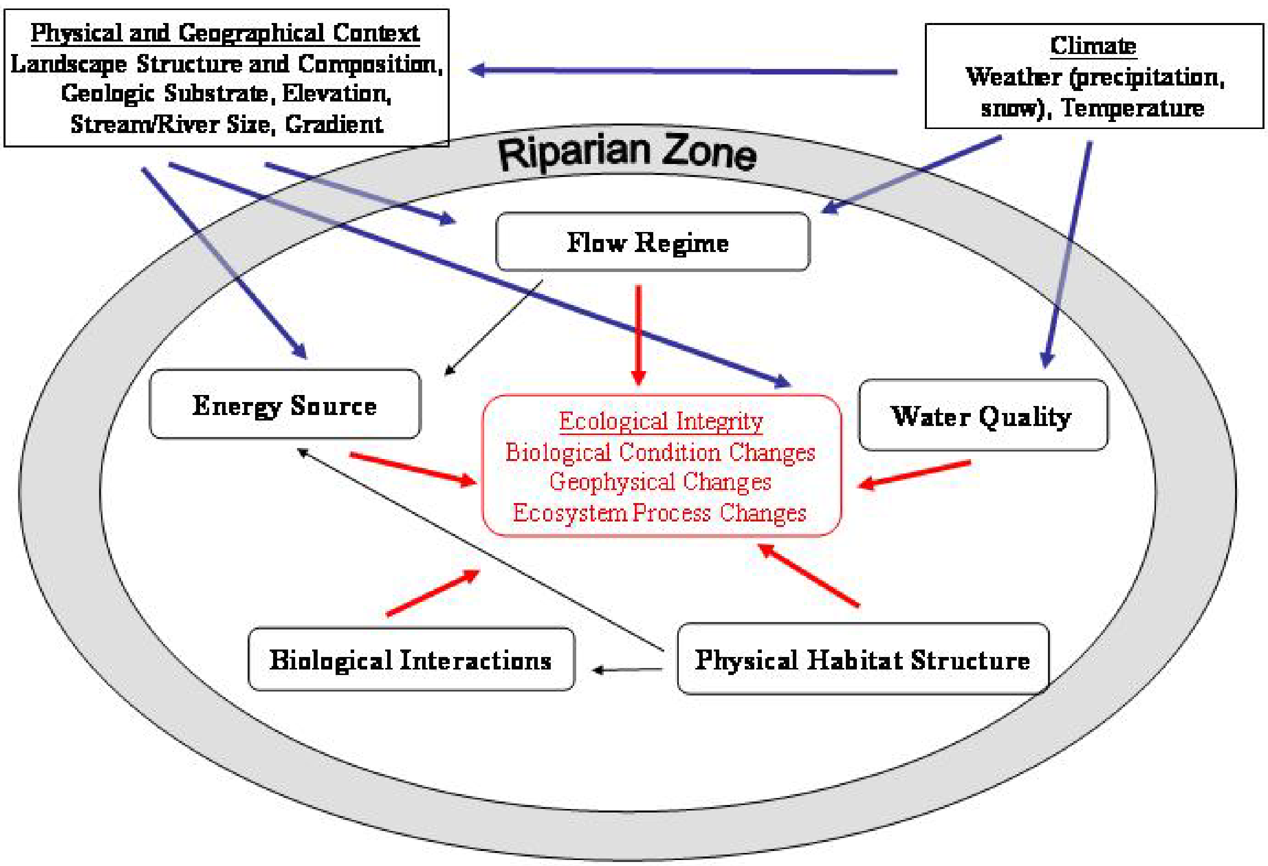


Figure 3. Basic conceptual model of a stream/river ecosystem and its elements (modified from Karr 1991, 1999).

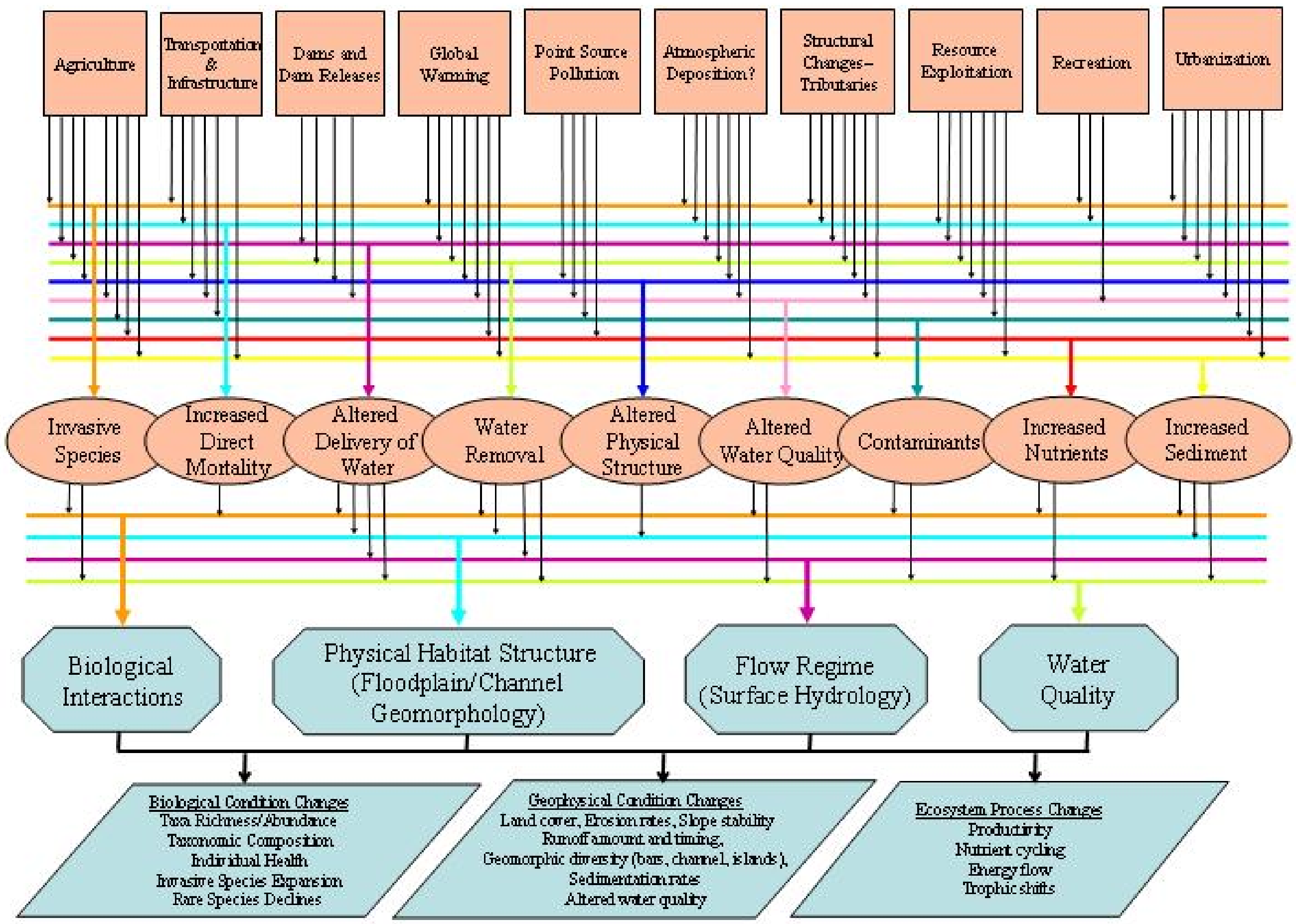


Figure 4. Tributary watershed anthropogenic drivers (rectangles), stressors (ovals), Karr's (1991, 1999) five ecosystem factors (octagons), and elements of ecological integrity. Connections (probable causal linkages) are represented by colored lines and thin vertical arrows.

Discussion of Model Elements

A primary goal of this paper is to synthesize the salient aspects of existing ecological theory as it relates to tributary systems (and their stressors) within the ERMN region. Thus, we are attempting to move toward a holistic understanding of the ecology and management of what could be termed "riparia", the areas of transition between water and land. We believe that it is useful to characterize tributary watersheds as a mosaic of interconnected hydrogeomorphic (HGM) patches or settings that contain a set of functional process zones (FPZs, Thorp et al. (in press)). FPZs consist not only of a distinguishable stream reach, but also include the geologic and topographic aspects of the surrounding terrestrial landscape. This can lead to considering streams, wetlands, and riparian areas as definable landscape units that support characteristic biota and that respond predictably to a set anthropogenic stressors. The formulation and discussion of such a conceptual model will further explain the critically important ecological functions and economic values provided by these aquatic ecosystems, and assist those concerned with their protection, conservation, and management in reaching their objectives.

The structure and function of these ecosystems including the influence of human activities and stressors is considered under four general headings corresponding to Karr's "five factors" (*Flow Regime, Energy Source, Water Quality, and Biological Interactions)*. The fifth factor, *Physical Habitat Structure* is addressed, in part, in each of the other sections. Finally, we include a section on landscape patterns and species movements to fully extend the framework to the watershed scale.

Flow Regime

Flow is a primary determinant of species composition in streams through its influence on carbon and nutrient transport (Newbold et al. 1982), habitat formation and stability (Giberson and Caissie 1998), and direct effects on species mortality patterns resulting from extreme flow events including floods and droughts (Reice 1984). While precipitation is the driving force in initiating a flooding event, the physical characteristics of the drainage basin, hydrology, and geomorphology of the stream-floodplain ecosystem are the primary factors controlling the concentration, spatial distribution, and dispersal rate of floodwaters (Staubitz and Sobashinski 1983, Scientific Assessment and Strategy Team 1994). Small streams are more influenced by precipitation events and are more unpredictable than larger rivers (Junk and Welcomme 1990, Benke et al. 2000). Although climate and geology are important, they are generally considered to be similar within a given region and wetland type. Differences in landscape-level characteristics, such as upland indicators of disturbance and stream size are important characteristics to consider. Site-level indicators, however, can be utilized when necessary since they tend to be sufficient predictors for functional assessments (Brinson et al. 1995). According to the Riparian Area Management's Proper Functioning Condition Workgroup (1993), riparian-wetland areas are functioning properly when site-level indicators such as adequate vegetation, landforms, or large woody debris are present to dissipate stream energy and improve floodwater retention and groundwater recharge.

The physical characteristics of floodplain wetlands are important for assessing the potential of an area to store and manage floodwaters. Wetlands reduce the amount of runoff that reaches the streams by storing runoff from adjoining areas (Demissie and Khan 1993). This desynchronizes water delivery to streams, which decreases the frequency and magnitude of flooding downstream (McAllister et al. 2000). Unobstructed floodplains provide a broad area for floodwaters to dissipate energy through the reductions of water velocities, flood peaks, and erosion. Floodplain vegetation retards water flow through surface roughness (Arcement and Schneider 1989) and small topographic depressions temporarily trap floodwater as long term storage (Owen and Wall 1989).

It is useful to consider the flow of water and materials from the upper reaches of the watershed to lower reaches. Initially, waters at the watershed boundary begin to accumulate in surface and near-surface areas. Precipitation, surface runoff, and near-surface runoff (i.e., interflow) accumulate in narrow, ephemeral or intermittent channels. A portion of the precipitation component infiltrates into shallow and deep aquifers. This is dependent on the areal extent, strata, and composition of vegetation, soil type, topographic gradients, surficial geology, and coverage of human-built structures. Discharges of shallow and deep ground water may be expressed at the surface as springs, seeps, and slope wetlands, or below the surface entering directly into streams and wetlands. Such discharges generally constitute the base flow to these aquatic systems. Eventually, somewhat dependent on season, sufficient water accumulates to sustain the flow in a perennial stream. Whereas the zero-order channel tends to dry out seasonally, first order streams tend to have a persistent base flow, usually in a relatively linear channel with little or no floodplain. These relatively small elements are strongly influenced by the characteristics of the adjacent riparian corridor, whether it be the amount of tree cover, type of soil, or range of stressors. These influences, separated from inputs originating upstream, can be referred to as lateral effects.

As flow increases, energy also increases to the point where physical modifications to the channel can occur. Pool-riffle complexes develop in the widening channels of tributary streams (second to fourth order)(Forman 1995). Floodplains continue to widen as the flow transitions from tributary streams to larger rivers. In these stages, the river itself, and to some extent the adjoining floodplain, are tied more closely to the characteristics and periodicity of the flows that have accumulated from upstream reaches, and less by the activities in the riparian corridor. Mid-reach and mainstem portions of the river network become uncoupled from upland hillslopes and the sediments eroded from uplands. Sediments, now deposited in the alluvial floodplain, define the channel and its flow path. This dependency is represented in Figs. 2a-c by the size of the arrows which represent the strength of influence.

When one incorporates components outside the stream channel proper into the tributary model, complexity of the ecosystem increases. The accumulation and flow of water across the landscape coupled with the varied microtopography of these areas results in a *river mosaic* of hydrologically-derived gradients and discontinuities across the surface (Forman 1995). The wetland components of this mosaic can be referred to as a *headwater complex* (D. Wardrop, pers. comm.). Previously, wetlands were classified primarily on the dominant vegetation and hydrology (Cowardin et al. 1979; used to code the National Wetlands Inventory). More recently, the hydrogeomorphic (HGM) approach (Brinson 1993, Smith et al. 1995) has provided additional elements for classifying wetlands (i.e., water source, water dynamics, landscape position) and for comparing functions and condition across reference sites. In tributary watersheds, the most relevant HGM subclasses of wetlands are headwater floodplains, riparian depressions, and slopes, all of which can contribute to a headwater complex, and by association, to a river mosaic (Cole et al. 1997, Brooks et al. In prep.).

Human activities upstream influence flood frequency and intensity (McAllister et al. 2000). Urbanization creates impervious surfaces and underground sewers, which accelerate the delivery rate of surface water to the stream (Pennsylvania Environmental Council 1973, Paul and Meyer 2001). As little as 3% impervious cover in a contributing area has been shown to negatively impact the ecological integrity of aquatic ecosystems (e.g., May et al. 1997). Serious declines in biotic integrity have been observed when urban land exceeds 7% of total watershed area (Synder et al 2003). Channelization, levees, and floodwalls both on-site and upstream destroy wetland and riparian habitat, restrict river flows, decrease water elevations at low flows and increase water levels at the same locations during floods (Scientific Assessment and Strategy Team

1994). Channelization funnels water into the stream, rather than allowing overbank flow to spread water across wetlands and decrease velocity (Brown 1988). This results in a decrease in the ability of wetlands to perform other functions, such as removing sediment and nutrients, and long-term surface water storage (Johnston et al. 1984, Brown 1988, Rheinhardt et al. 1999) and altering stream morphometry which leads to scouring and incision. Highway embankments remove vegetation, eliminate natural storage areas, and reduce space available for floodwater storage (Owen and Wall 1989). These and other activities often result in channel degradation, which lessens the depth, frequency, duration, and predictability of flooding. The floodplain frequently becomes isolated from the stream channel through incision and no longer has the opportunity to perform this function. These activities not only impair the performance on-site, but they also increase the flood pulse downstream, a process that places additional pressure on downstream wetlands to dissipate energy and temporarily detain floodwaters.

Changes in the structure and composition of surrounding forests can also have large effects on stream flow. For example, complete removal of forest vegetation associated with logging dramatically increases annual water yields and bank flow flood frequencies (reference). In addition, human or pest-induced changes to the composition of surrounding forests can alter stream flow. For example, the hemlock woolly adelgid (HWA), an exotic insect forest pest that kills eastern hemlock trees has been identified in numerous component parks in ERMN. The pest is expected to cause significant, and perhaps complete hemlock mortality. In a study designed to determine potential effects of HWA-induced hemlock decline on headwater streams in DEWA, Snyder et al. (2002) found that headwater streams draining hemlock forests were less likely to dry up completely during drought years than similar streams draining mixed hardwood forests. Based on their findings they predicted that HWA will have dramatic effects on headwater stream biodiversity.

Long-term surface water storage helps to maintain the characteristic hydroperiod of wetlands and streams. Hydroperiod affects just about all components of aquatic ecosystems; plant communities, soil processes, nutrient cycling and faunal communities are all influenced by the duration and frequency of inundation (Gosselink and Turner 1978, Carter 1986, Tiner 1998). Standard gauging stations have long been used to plot the expected hydrographs for streams and rivers throughout the U.S. These data are readily available digitally, although not all streams are gauged. On a smaller scale, the Penn State Cooperative Wetlands Center has prepared typical hydrographs of the expected hydrologic regime for making comparisons among wetland subclasses (Figure 5)(Brooks 2004, Cole and Brooks 2000, Cole unpublished). Deviations from this expected pattern can be used to suggest the presence of watershed stressors.

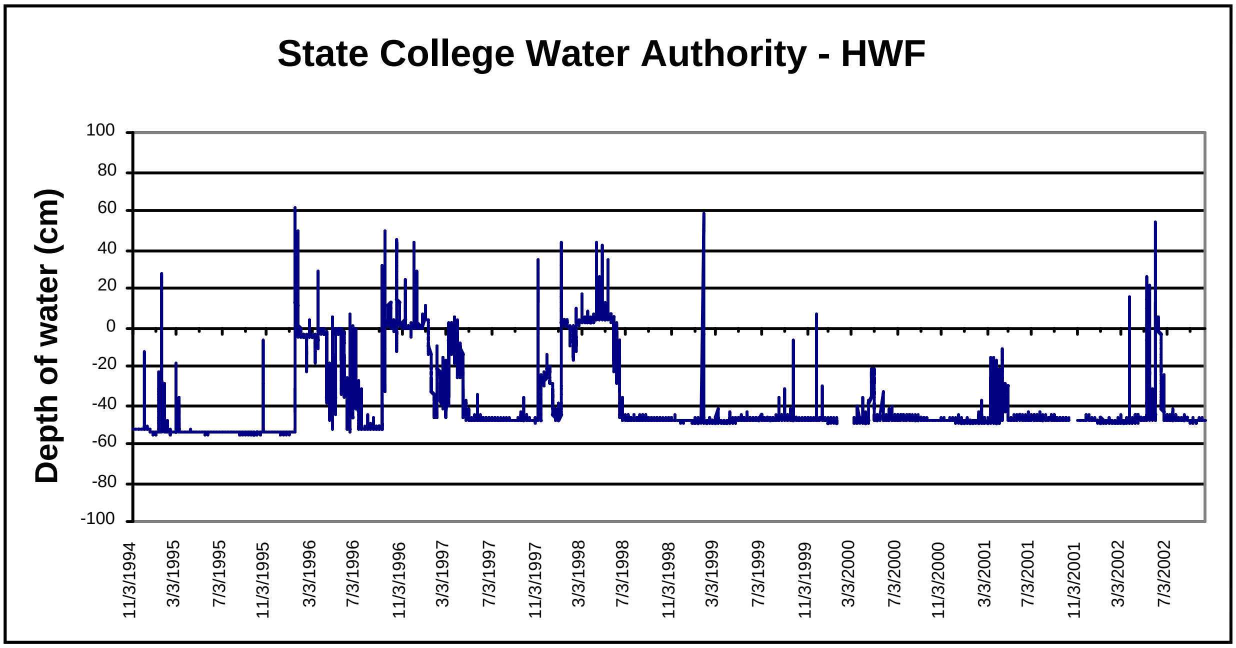


Figure 5. Typical hydrograph for HGM Wetland Subclass - Headwater Floodplain.

The amount of flooding in a wetland or floodplain is dependent on climate, topography, channel capacity and slope, soil and lithology (Novitski 1989, Brinson 1990). Physical characteristics of a site determine the ability of a wetland or floodplain to retain this excess water. The presence of macrotopographic depressions, whether hydric or not, affects the potential of a site to retain incoming waters for long periods of time. Features such as oxbows, meander scrolls, and backswamps all constitute macrotopographic depressions (Brinson et al. 1995). Various impediments to flow may reduce the storage function of the hydric (e.g., wetlands and auxiliary stream channels) and upland (e.g., natural levees, non-hydric inclusions) components of floodplains. Channelization increases the rate of runoff, which increases peak flow, and decreases water storage and the residence time of water (Brown 1988). Studies show that increases in water level fluctuation relate directly to increases in runoff from adjacent uplands (Euliss and Mushet 1996). Human alterations also cause an increase in the amount of sediment transported in a stream and ultimately across a floodplain. Excess sediment can fill critically important interstitial spaces in the substrate of streams, reducing or eliminating aquatic biota (e.g., larvae of aquatic insects, salamanders, and fishes). This same source of sediment may result in the filling of depressions, and hence a reduction in the storage capacity and topographic complexity of wetlands and the floodplain in general.

Energy Source

The changes in the relative importance of energy sources and the associated changes in plant and animal species structure and composition is the basis of the River Continuum Concept (RCC)(Vannote et al. 1980). Essentially, the RCC proposes that, in unperturbed watersheds, stream communities change in predictable ways as we move from headwaters to large rivers and these changes are mediated in large measure by the amount and sources of energy. Within the headwater stream component of tributary watersheds (0 through 2nd order), the source of energy is mainly from outside the stream channel (i.e., allochthonous inputs), largely in the form of leaf litter, or coarse particulate organic matter (CPOM)(Cummins et al. 1973). The quantity, quality, and timing of leaf litter inputs vary depending on regional climate and the structure and composition of the surrounding forests. In watersheds draining deciduous forests (which predominate in ERMN tributary watersheds), the majority of leaf litter inputs to streams occur in autumn when deciduous trees naturally loose their leaves. However, in watersheds comprised mainly of coniferous forests, inputs may be more consistent throughout the year. Once leaves or conifer needles fall into headwater streams, a large fraction of the associated carbon is rapidly dissolved or leached directly into the water column and transported to downstream reaches with flow. The remaining CPOM tends to accumulate in pools and stream margins into leaf packs where they are colonized and fed upon by bacteria and aquatic fungi, a process termed conditioning (Cummins and Klug 1979). Conditioned leaves are then available as food for aquatic macroinvertebrates which in turn supports the production of fish and other secondary consumers. In addition to being used directly by microbial and macroinvertebrate assemblages, a significant fraction of the CPOM component is broken down into smaller particles or fine particulate organic matter (FPOM), by the abrasive forces of stream flow and by the feeding activity of leaf-shredding macroinvertebrates (Boling et al. 1975, Iversen et al. 1982). Subsequently, FPOM is suspended into the water column and exported to stream reaches downstream.

In forested watersheds, the quantity of leaf litter that enters streams is not limiting. However, the extent to which litter inputs are available to stream communities depends on two factors. The first is extent to which headwater streams can retain CPOM within headwater reaches in face of downstream flow. Although increases in flow associated with storms are responsible for most export of CPOM from headwater streams on an annual basis (Schlesinger and Melack 1981), correlations between discharge and transport of organic carbon are weak in headwater streams, especially during non-storm periods (Cuffney and Wallace 1989) indicating the importance of retention. There are many factors that influence organic matter retention including biological uptake. However, the physical structure and complexity of stream channels have been implicated as primary determinants. Specifically, physical features of streams such as boulders and a stream channel that allows floodwaters to overflow their banks, slow the transport of water and materials downstream. Of particular note, is the role that coarse woody debris (CWD) plays in retaining particulate organic matter within headwater stream reaches (e.g., Bilby and Likens 1980, Wallace et al. 1995, Brookshire and Dwire 2003). Recent research indicates that undisturbed watersheds contain more CWD and are more retentive than disturbed watersheds thus enhancing the availability of organic material to benthic consumers (Wallace et al. 2001, Scott et al. 2002).

The second factor that affects organic matter availability in headwater streams is the species composition of the leaf litter itself. Specifically, leaves of different plant species break down at different rates due in large measure to the chemical characteristics, especially nitrogen and fiber content, of the leaves (Webster and Benfield 1986). In an extensive study of leaf breakdown in streams, Peterson and Cummins (1974) found wide variation in leaf breakdown rates among species and suggested that this variation ensured that carbon was available to secondary consumers throughout the year. Conifer needles break down much slower than deciduous species and the leaves of herbaceous plant break down faster than those of woody plants. Thus, disturbances that change the composition of the surrounding riparian area would be expected to result in changes in the amount and timing of organic carbon available to stream communities. Forest pests such as hemlock woolly adelgid and gypsy moth are both common in several ERMN parks and can have dramatic effects on the species composition of riparian forests. In addition, numerous abiotic factors have been shown to affect litter decomposition rates within plant species. For instance, litter breakdown rates are positively correlated with temperature and dissolved nutrients, and negatively correlated with acidity and various toxic effluents (reviewed in Webster and Benfield 1986). Consequently, factors that reduce water quality are also expected to significantly alter the energy pathway in headwater streams and lead to disruptions in ecological integrity. Water quality of tributary resources is a major concern in ERMN.

Further downstream in the mid-reaches (3rd and 4th order), stream channels begin to widen which allows more light to penetrate the forest canopy and reach the stream bottom. At this point, instream primary production (i.e., autochthonous inputs) becomes an important energy source mainly in the form of benthic brown algae (i.e., diatoms) (Molloy 1992). In addition, FPOM derived and exported from upstream reaches also represents a significant energy source to stream biota. Thus, in unperturbed mid-reaches, direct litter inputs from the riparian zone diminish in importance and instream primary production and carbon inputs derived from upstream reaches become more important carbon sources to fuel secondary production.

In addition to increased production and diversity of benthic algae in mid-reach streams, the composition of macroinvertebrate assemblages also change in response to changing sources of energy. In headwater streams where energy is mainly in the form of leaf litter inputs from the surrounding riparian forest, macroinvertebrate assemblages are dominated by taxa specialized to feed on CPOM (i.e., "shredders"). In contrast, in mid-reach streams, taxa specialized to feed on benthic algae (i.e., "grazers") replace the shredder group (Vannote 1980). In addition, organisms specialized to feed on FPOM that is either suspended in the water column ("filter-feeders") or settled out on stream bottoms ("collector-gatherers") also become more important in mid-reach streams to take advantage of FPOM generated and exported from headwater reaches (Vannote 1980).

Therefore, from an energy perspective, the ecological integrity of stream communities in mid-reach streams is determined mostly by factors that affect retention, transport, and the quality of organic matter from headwater areas upstream, and by factors that influence instream primary production within mid-reach areas. In particular, the effects of non-point source pollutants associated with agriculture and urban land use in upstream or adjacent landscapes (both of which are significant concerns in ERMN) have been shown to affect energy pathways in mid-reach areas. Herbicides and increased sediment inputs have been shown to reduce overall instream primary production with subsequent changes in macroinvertebrate diversity and production (Georgian and Wallace 1983, Guasch et al. 1998). Also, nutrient enrichment from agriculture has been shown to cause a shift in benthic algal composition from an assemblage dominated by diatoms, a preferred food source of many macroinvertebrate species, to an assemblage dominated by filamentous green and blue-green algaes that detritivores mostly avoid (Hart and Robinson

1990, Jacoby et al. 1990). Acidification of stream habitats has also been shown to alter primary production in streams (e.g., Planas and Moreau 1986).

The RCC also makes specific predictions regarding energy sources in larger rivers and the associated responses of biological communities. However, we do not consider them here in light of our focus on the tributary component of river networks. Moreover, the data from larger rivers show less agreement with the predictions of the RCC than do headwater and mid-reach streams (Thorpe et al, in press). Nevertheless, the importance of the ecological integrity of tributary watersheds to the structure and function of larger rivers is indisputable. Therefore, it seems to us that it is in the interest of the “large river” group to work closely with the tributary group in developing vital signs that represent the ecological integrity of entire riverine systems.

Tributary components of headwater systems are mainly forested under relatively unaltered conditions throughout the ERMN region. Not only do trees contribute leaf litter and downed wood described below, but forests provide shade, root structure for stream bank stability, and other physical and microclimatic controls to the stream. Wetlands in the riparian zone maintain the organic-rich conditions in the soil that is critical for denitrification in groundwater flowing to stream channels from nitrate sources such as agriculture. Forested wetlands support both aquatic and terrestrial food webs: aquatic food webs when they are flooded and terrestrial food webs when they are seasonally dry. As in nearly all terrestrial and aquatic food webs, the detrital food web dominates over grazing pathways (Brinson et al. 1981).

What sets apart headwater streams is the extent to which they are hydrologically connected to riparian wetlands. In contrast to larger stream orders, where large volumes of water flow past floodplain wetlands, both headwater streams and their associate wetlands are minute by comparison, but they are everywhere, often at a density of 1km of stream per 1 km^2 of land surface. The consequence of this proximity and abundance is that they are most exposed to human activities that modify wetland condition. The most pernicious is channelization (straightening, deepening, and widening), a process that removes most hydrological and biological connections between stream channels and floodplain. The conversion of riparian forest to agriculture, pasture, residential areas, and even urban land uses fundamentally changes all ecological processes. To the extent that overbank flow during floods connected stream habitat with floodplain habitat, channelization totally disrupts this connection and simplifies the complexity of food webs and the complex pathways of energy. Some land uses that follow channelization mentioned above alters habitat conditions so fundamentally for forest species that they cease to exist.

Detrital biomass is an important component of headwater ecosystems and plays a role in nutrient cycling and habitat for plant and animal communities in tributary watersheds. Detrital biomass is represented by snags, down and dead woody debris, organic debris on the forest floor, and organic components of mineral soil. This has been described for wetlands in the national riverine HGM model (Brinson et al. 1995) and regional HGM models (Brooks 2004), and for Mid-Atlantic streams by Barbour et al. (1999) and Boward et al. (1999). Detritus is considered an indicator of the potential decomposition and nutrient cycling rates at a site. Decomposition is generally faster in aquatic than terrestrial landscapes due to increased leaching, fragmentation and microbial activity (Shure et al. 1986). Large pieces of coarse woody debris (CWD) derived from adjacent or upstream forests are processed into fine particulate organic matter (FPOM) and then further processed and incorporated into organic matter (Bilby and Likens 1979, Jones and Smock 1991). Organic material may be transported to channels or respired as CO_2 at any stage of the decomposition process (Bilby and Likens 1979, Jones and Smock 1991).

Wetlands are a major source of particulate organic carbon (POC) entering streams. Woody debris is a nutritional substrate, provides habitat for microbes, invertebrates, and vertebrates, is a substrate for seedling growth, and serves as a long-term nutrient reservoir; a consistent source of organic material (Harmon et al. 1986, Brown 1990). POC is a small fraction of total organic carbon (TOC), but ranks disproportionately higher as a food source for fish and invertebrates (Taylor et al. 1990). POC from wetlands contributes substantial amounts of organic matter to stream channels (Muholland and Kuenzler 1979, Dosskey and Bertsch 1994). In fact, POC comprises between 24% and 46% of the total organic carbon in streams (Dosskey and Bertsch 1994). Detrital inputs to the stream during peak inundation periods support microbial and macroinvertebrate communities in the stream channel (Smock 1990). The rate of particulate matter degradation depends on many factors, including soil moisture levels. According to Bilby et al. (1999), when compared to either fully submerged or terrestrial conditions, wood decays at a much faster rate when periodically wetted and dried, conditions typical of many wetlands and floodplains. Floodplains had higher decomposition rates for wood than streams (Cuffney 1988). Forested riparian corridors maintain more benthic habitat, increase channel and bank stability, and provide additional contact area for transforming both nutrients and pesticides than non-forested reaches (Sweeney et al. 2004).

Water Quality

Headwater stream and wetland communities are strongly influenced by the chemistry of the water (Jones and Mulholland 2000). Natural variation in water hardness, specific conductance, acidity, and dissolved oxygen are all major determinant of species composition, and consequently must be considered when designing a sampling program to monitoring aquatic resources. Human sources of pollution reduce water quality and alter aquatic communities directly by killing or weakening individuals, or by altering energy pathways.

In tributary watersheds, measures of water chemistry are more reflective of the geologic and topographic characteristics of the landscape than for larger rivers. The complex geology of the Appalachians can create circumstances where relatively short stream reaches and individual wetlands can have a different water chemistry than their neighbors (USEPA 2000, Synder et al. In review). Such variability produces extraordinary biodiversity.

In ERMN component parks, numerous human sources of water quality degradation have been identified including urbanization, failing septic systems, agriculture, acid mine drainage, and acid precipitation. Wetlands and riparian corridors often act as buffers to these water sources due to their ability to filter out and transform contaminants (e.g., Figures 2a-c). Of particular importance to ERMN parks is the introduction of non-point pollutants including nutrients such as nitrogen, phosphorus, pesticides, herbicides, and sediments that enter stream and wetland habitats through groundwater and surface runoff.

Eutrophication from excess nutrients (e.g., nitrogen and phosphorus) can be a significant stressor in tributary watersheds. Over time, eutrophication typically alters energy pathways by increasing primary production (see section on "energy flow" above) which often results in lower dissolved oxygen concentrations. These changes usually lead to highly productive, but taxonomically and trophically simple biological communities in both streams and wetlands (Sandin and Johnson 2000, Brinson and Malvarez 2002). Herbicides also disrupt energy pathways, but they cause reductions in instream primary production, and pesticides directly affect survival and reproduction of populations. Excess turbidity caused by high levels of suspended sediment decreases oxygen levels and photosynthesis rates, impairs the respiration and feeding of aquatic organisms, destroys fish habitat, and kills benthic organisms (Johnston 1993b). In wetlands, high sedimentation rates decrease the germination of many wetland plant species by eliminating light penetration to seeds, lowering plant productivity by creating stressful conditions, and slowing decomposition rates by burying plant material (Jurik et al. 1994, Vargo et al. 1998, Wardrop and Brooks 1998, Mahaney et al. 2004).

In some instances wetland and riparian habitats can be effective mitigators of non-point source pollutants, especially nutrients and sediments, due to their ability to filter and transform contaminants (Figures 2a-c). Because sediments and phosphorus are transported from uplands to streams and wetlands through surface flow (phosphorus largely attached to sediment particles) (Lowrance et al 1984, Pionke et al. 1986), the primary removal mechanisms for phosphorus and metals are the settling of particles out of the water column and adsorption to organic matter and clay. Long-term removal can occur through roots, buried leaves, and sediment deposition (Richardson and Craft 1993). As long as there is sufficient time for transported material to come in contact with surface litter, riparian vegetation can be effective in retaining sediments and nutrients. For example, in a floodplain wetland in Sweden, 95% of phosphorus entering the wetland in surface runoff was removed within 16 m (Vought et al. 1994). In North Carolina, approximately 50% of the phosphorus leaving agricultural fields in runoff was removed in riparian areas (Cooper and Gilliam 1987). However, during storms and in high-gradient watersheds, sediment retention by riparian zones is less effective (Jordon 1986). Phosphorus is even more sensitive to flow rates because it tends to bind to smaller particles that are less efficiently trapped by surface litter.

In contrast, nitrogen moves primarily through ground water as dissolved nitrate, ammonia or organic nitrogen (Peterjohn and Correl 1984, 1986). Most nitrogen is removed from subsurface water through denitrification by soil microbes within wetlands and riparian soils (Davidsson and Stahl 2000). Research has shown that riparian forests are capable of retaining up to 89% as compared to 8% for cropland, and the nitrogen loss from the forest was primarily via groundwater (Peterjohn and Correll 1984, Gilliam 1994, Jordan et al. 1997). But as with sediments and phosphorus, retention of nitrogen is also more efficient at low discharge (Schnabel 1986). During high discharge relatively more water moves from upland and riparian areas to streams and lowland wetlands through surface flow. Thus, there is less time for vegetative uptake and microbial transformation of nutrients (Pionke et al. 1986). Research has shown a 90% or more reduction in NO_3^- concentrations in water as it flows through riparian areas (Gilliam 1994). Organic matter is also important in providing a substrate necessary for microbes to perform the process of denitrification. Plant uptake is an additional means of nitrogen removal from the system.

Sediment retention in wetlands and riparian corridors not only removes phosphorus, but has the additional function of reducing turbidity and contaminants sorbed to sediments, thus benefiting neighboring streams, rivers, and lakes (Oschwald 1972, Boto and W. H. Patrick 1978, Cooper and Gilliam 1987, Hemond and Benoit 1988, Johnston 1991). While wetlands and floodplains have been shown to trap sediment in relatively unaltered settings, accelerated sedimentation can quickly overwhelm the capacity of these habitats to store and process the sediments (Jurik et al. 1994, Wardrop and Brooks 1998, Freeland et al. 1999). High sedimentation rates decrease the germination of many wetland and riparian plant species by eliminating light penetration to seeds, lower plant productivity by creating stressful conditions, and slows decomposition rates by burying plant material (Jurik et al. 1994, Vargo et al. 1998, Wardrop and Brooks 1998, Mahaney et al. 2004). Excess turbidity caused by high levels of suspended sediment decreases photosynthesis rates, which may depress oxygen levels, thus affecting the respiration and feeding of aquatic organisms, destroying fish habitat, and causing outright death of benthic organisms (Johnston 1993b).

Landscape disturbances impact sediment loading and retention within the aquatic components of tributary watersheds. Hupp et al. (1993) found sedimentation rates to be highest in wetlands located downstream from agricultural and urban areas. Phillips (1989) found that between 14% and 58% of eroded upland sediment is stored in alluvial wetlands and other aquatic environments. As much as 90% of eroded agricultural soil was retained in a forested floodplain in North Carolina (Gilliam 1994). Eighty-eight percent of the sediment leaving agricultural fields over the last 20 years was retained in the watershed of a North Carolina swamp (Cooper et al. 1986). Approximately 80% of this was retained in riparian areas above the swamp and 22% was retained in the wetland itself.

Another major threat to water quality in several ERMN parks is increased acidity associated with acid mine drainage (AMD) and acid deposition (AD). Numerous component parks have streams impacted by AMD including ALPO, JOFL, FRHI, NERI, GARI, and BLUE (Marshall et al. 2004). In addition, acid deposition is likely a problem in several parks including DEWA and NERI. Snyder et al. (2005) found that over one quarter of wetlands and vernal ponds in DEWA had pH levels <5 and all of these failed to support amphibian breeding populations that were common at sites with more buffered pH. Moreover, as a region, the pH of rainfall in the area of North America that contains all of the ERMN parks is among the lowest nationwide (NADP 2003).

Increased acidity can have dramatic effects on stream and wetland communities. Increased H^+ ions directly disrupt ion regulation in most animal species causing death or compromising fitness depending on the level (Gerhardt 1993). Certain metals such as aluminum, that are prevalent but relatively inert in streamside soils and stream sediments, become dissolved, mobilized, and toxic to aquatic species at low pH (Nelson and Campbell 1991). In addition, when acidic waters merge with pH neutral or basic waters at stream junctures, certain metal complexes such as iron hydroxide precipitate out of solution and coat stream substrates thus smothering benthic algae and macroinvertebrates (Hoehn and Sizemore 1977, DeNicola and Stapleton 1992). Consequently, acid effects can extend downstream even in areas where stream pH is relatively high. Finally, leaf litter decomposition rates in headwater streams and wetlands are significantly reduced as streams become acidified (Kittle et al. 1995, Niyogi et al. 2001).

In the case of AMD, acidity and metal concentrations are frequently so high that the effected stream or wetland may be devoid of all life. In less extreme cases, AMD and AD has been shown to adversely affect the species diversity and productivity of benthic algae (e.g., Verb and Vis 2000), macroinvertebrates (e.g., Rosemund et al. 1992), and fish (Carline et al. 1992). The pH of water in streams receiving AMD or AD is often poorly correlated with the pH of the sources indicating that some systems are more vulnerable than others to acidification. As mentioned above, the water chemistry of headwater streams are more strongly related to the geology and terrain of the surrounding watershed than for larger rivers. One important characteristic of headwater stream water chemistry that is an important determinant of sensitivity to AMD and AD is the concentration of base cations (e.g., calcium and magnesium). Streams that have high base cation concentrations typically have high acid neutralizing capacities (ANC), and are therefore, more able to maintain a stable pH despite AD (Faust 1983). In headwater streams, base cation concentrations are largely a function of the underlying surface geology. Streams underlain by carbonate geologies such as limestone supply considerable ANC to streams compared to geologies with little or no base cations like sandstone. However, the ability of carbonate geologies to buffer acidity associated with AD also depends on the amount of time that streams are exposed to AD. Specifically, in streams exposed to AD, the production of base cations through mineral weathering is slower than the rate they are leached into the stream. Thus, over time, the pool of available cations may become depleted causing a threshold effect whereby the ability of carbonate geologies to buffer AD is compromised (Kirchner 1992).

Wetlands also appear to offer some mitigation potential for acidified streams. For example, comparative research studies have shown that beaver ponds generate significant ANC to associated streams resulting in more stable pH (Cirmo et al. 2000, Margolis et al. 2001). Moreover, laboratory experiments using simulated wetlands have demonstrated that wetland soils act as sinks for strong acid anions (nitrates and sulfates), and wetland microbial communities transform toxic metals to less toxic or available forms (Williams et al. 1994). Constructed wetlands have been shown to be an effective mitigation tool for restoring streams affected by AMD removing up to 99% of the iron and aluminum and up to 30% of the nitrogen loading (Brenner 2000).

Biological Interactions

The biological diversity of tributary watersheds in the Mid-Atlantic region has been documented reasonably well. Some taxa pertinent to the region are particularly diverse, notably salamanders, freshwater mussels, aquatic insects, and breeding neotropical migrant songbirds (e.g., Stein et al. 2000). Various investigations have tallied the species and communities that are prevalent in the region (e.g., Majumdar et al. 1989, Croonquist and Brooks 1993, Snyder et al. 2002, Ross et al. 2003).

The maintenance of a characteristic plant community is a designated hydrogeomorphic (HGM) function for wetlands that also relates to a variety of ecological functions in tributary watersheds such as: energy dissipation via roughness, detrital production and nutrient cycling, and biodiversity and habitat functions. The composition of vascular plant communities has long been used to characterize wetlands (Cowardin et al. 1979, Tiner 1988, Mitsch and Gosselink 2000). Plant community composition influences many ecosystem properties, such as primary productivity, nutrient cycling and hydrology (Hobbie 1992, Ainslie et al. 1999). Plant species composition plays an important role in determining soil fertility (Wedin and Tilman 1990, Hobbie 1992). Individual plant species effects on ecosystem fertility can be as important, or more important, than abiotic factors, such as climate (Hobbie 1992). Plant community composition also influences the habitat quality for invertebrate, vertebrate, and microbial communities in both wetlands and streams (Gregory et al. 1991, Norokorpi 1997, Ainslie et al. 1999).

Plant communities may be highly modified by human alterations that facilitate colonization by invasive and aggressive species. Invasive species change competitive interactions, which result in changes in species composition (Walker and Smith 1997, Woods 1997). A checklist, which includes provisions for invasive plants, has been developed to record any observed stressors on streams, wetlands, and riparian areas in the region (Brooks 2004). Streams and riparian systems are particularly vulnerable to exotics because their linear nature exposes them to invasions (Simberloff et al. 2005).

Tributary streams are important for selected fisheries (e.g., salmonids), but in general, do not support the high biomass or species richness present in larger rivers. The distribution and habitat of fish species was documented by Cooper (1983) and others for the region, and there have been fish inventories in individual park components (e.g., Leonard and Orth year; Ross year). In high gradient headwater streams, brook trout, various minnows (Cyprinidae), and sculpins (Cottidae) are common. Boltz and Stauffer (1989) highlighted the fishes that are dependent in some manner on wetlands and their connectivity with streams. Although the richness and abundance of fish in tributary watersheds can be a useful indicator of condition, fish penetration into the upper reaches of these ecosystems is limited (Church 2002). In places where fish are not present in abundance, amphibians, particularly streamside salamanders, and riparian birds can serve as alternate vertebrate indicators (Brooks et al. 1998, O'Connell et al. 2003, Rocco et al. 2004).

The importance of the wetland and riparian components of tributary watersheds as habitat for wildlife communities is reasonably well documented in the Mid-Atlantic region. Just as rivers and lakes provide fisheries habitat, the provision of wildlife habitat is an oft cited function of the adjacent wetlands and riparian areas. Profiles for various taxa are summarized in Majumdar et al. (1989), Brooks et al. (1993), and Tiner (1998). Obligate and facultative fauna using these stream, wetland, or riparian habitats can include seasonal (e.g., aquatic insects, winter migrant birds, summer foraging bats), resident (e.g., freshwater mussels, cyprinid minnows, salmonids, streamside salamanders, beaver), wide-ranging (e.g., mink river otter, herons), or breeding migrant (e.g., belted kingfisher, Louisiana waterthrush, Acadian flycatcher) species.

Fisheries, as an important commercial and recreational resource, are commonly monitored in streams and rivers by resource agencies, yet, we seldom have resources to census the correspondingly diverse wildlife community. A commonly used alternative is to assess *potential* wildlife use with Habitat Suitability Index (HSI) models (USFWS 1980, Morrison et al. 1992, Anderson and Gutzwiller 1994). HSI models have been used as a means to estimate the level of wetland functioning as wildlife habitat based on consistent use of 10 common species (Brooks and Prosser 1995, Brooks 2004). A similar group of 10 common vertebrates has been proposed for assessing the condition of stream and riparian corridors (Brooks unpublished).

Landscape Patterns and Species Movements

As emphasized throughout this document, tributary resources are a collection of wetland, riparian, and stream habitats connected by the movement of water, carbon, and nutrients. However, species also move within and among various habitats and, consequently, the biological integrity of a given area also depends on factors that affect species movement. Within the stream network itself, fish and aquatic macroinvertebrates use different habitats at different times of the day, year, and phases of their life cycle. For example, aquatic macroinvertebrates move downstream with the water column, a process known as "drift". Invertebrate drift rates have been shown to have a diel periodicity, with higher rates at night and peaks near dusk and dawn (Waters 1965). Vertebrate predators have been shown to respond to these diel drift patterns (Griffith 1974, Hughes 1998). Drift has been classified as active or passive depending on whether species intentionally enter the drift as a dispersal mechanism in response to food availability or predation risk, or accidentally with flow (Allan 1995). However, although the relative importance of these two types of drift is still debated, the evidence is clear that drift is an important process in headwater streams and larger rivers. For example, the cumulative number of drifting insects over a reach of stream in a 24-hour period could be as much as 100 times benthic density (Waters 1972).

Obviously, any physical barrier to downstream movement would affect drift. Natural barriers such as waterfalls and beaver dams, and artificial barriers such as human constructed dams have been shown to affect invertebrate drift rates (Radford and Hartland-Rowe 1971, Schlosser 1998), and are both important features in several ERMN parks. DEWA alone has over 200 dams located on small streams, and both waterfalls and beaver dams are common. Moreover, invertebrate drift rates have been shown to correlate with increases in flow (Bosco and Perry 2000) and droughts (Cuffney and Wallace 1989), water temperature (Dudgeon 1990), light levels (Anderson 1966) and water quality, including stream acidification (Courtney and Clements 1998), heavy metal pollution (Beltman et al. 1999), and pesticides (Wallace et al. 1987). Invertebrate drift rates have also been shown to increase following removal of CWD from the stream channel, probably due to effects on flow or food availability (Ellliott 1986). Thus, in addition to the obvious effects of physical barriers, any disturbance that alters flow, temperature, food availability, or water quality would also be expected to alter downstream movement of aquatic invertebrates.

Fish also move longitudinally within the stream network. The most obvious examples are taxa that migrate upstream to breed, including many species of salmonids (trout) and catastomids (suckers). However, because all fish vary dramatically in size from embryo to adult, most species exhibit complex life cycles and habitat use patterns over the length of their life cycles that are mediated by migration (Schlosser 1995a). Moreover, many fish species have diel and seasonal migratory behaviors in response to food availability and natural variation in temperature and flow (Albanese et al. 2004). The size and distribution of mesohabitats (riffles and pools) within the stream channel have also shown to be important determinants of short-term fish movement patterns in headwater streams (Lonzarich et al. 2000).

Not surprisingly then, like invertebrate drift, fish migration is directly affected by natural barriers like waterfalls (Northcote and Hartman 1988) and beaver dams (Schlosser 1995b), as well as human created barriers like hydroelectric dams (Radford and Hartland-Rowe 1971). However, perhaps more important to ERMN tributary resources, are increases in urban and agriculture land uses in upstream portions of the watershed that have been shown to indirectly influence fish migration patterns through their effects on stream flow, temperature, water quality and distributions of stream habitat including the availability of hydrologic and thermal refugia (Schlosser 1995a, Pollino et al. 2004).

Lateral movements of species among upland, riparian, wetland and stream habitats are equally important in tributary watersheds. Aquatic insects, by far the most important component of the macroinvertebrate community in streams in terms of diversity and productivity, spend most of their life cycles in the stream environment. However, most species emerge from the stream as winged adults to mate and lay eggs, mostly in large mating swarms. The biomass of aquatic insects emerging from streams represents a significant energy source for riparian birds, mammals, and spiders and, therefore, represents a return of a significant amount of energy from the stream, back to the riparian area in which it was originally derived (Jackson and Fisher 1986). In addition, dispersal and oviposition of winged adults is the principal route of recolonization of streams denuded by floods, droughts, or pollution (Sheldon 1984). Although there has been relatively little research into factors influencing adult dispersal of aquatic insects, some studies suggest that the amount and composition of riparian forests are important determinants of dispersal distance and colonization success (e.g., Collier and Smith 1998, Briers et al. 2002). Thus, disturbances that alter the structure and composition of riparian forests would be expected to reduce the capacity or rate in which headwater streams recover from additional natural or anthropogenic disturbances.

Semi-aquatic species are also strongly influenced by the amounts, conditions, and spatial relations between upland, riparian, and stream and wetland habitats. In particular, most amphibian species require both terrestrial and aquatic habitats at various times of their life cycles. Some regionally important amphibian taxa such as several species of mole salamanders (*Ambystoma spp.)* and some anurans such as wood frogs (*Rana sylvatica*) spend most of their lives in terrestrial habitats but use vernal ponds and wetlands for breeding and larval nursery habitat (Semlitsch 2000). Other species such as the red-spotted newt (*Notophthalmus viridescens*) spend most of their adult lives in aquatic habitats but spend 1-2 years in terrestrial habitats as immature efts (Forester and Lykens 1991). As a result of this biphasic life history, amphibians depend on relatively undegraded terrestrial and aquatic components of the ecosystem to complete their life cycles. Moreover, the integrity of migration routes among habitats is critical in maintaining viable populations. The conversion of forest habitats to agriculture or urbanized landscapes, as well as increased development of roads have all been shown to disrupt dispersal and migration corridors of amphibians (Gibbs 1998, Joly et al. 2001, Guerry and Hunter 2002). Snyder et al. (2005) found that pond use by all three species of mole salamander found in DEWA was negatively correlated with primary roads.

In addition to aquatic and terrestrial habitat quality and intact migration routes, the size and isolation of breeding habitats have also been shown to be important landscape characteristics to amphibians. In contrast to most other faunal groups, pond-breeding amphibians do not show a positive relationship between habitat size and assemblage diversity. That is, smaller wetlands and ponds are disproportionately important to this group of animals because they are typically ephemeral and consequently do not support fish and other vertebrates that prey on amphibian larvae (Snodgrass et al. 2000). In addition, these abundant, small wetlands can function as stepping stones for dispersal and recolonization of extinct populations (Semlitsch and Bodie 1998). Despite their importance, small wetlands are usually more vulnerable to filling and draining associated with development because they typically lack state or federal regulatory protections (Semlitsch and Bodie 1998). Beyond direct habitat destruction, wetlands in general, and small wetlands in particular, are sensitive to changes in weather patterns. Therefore, the warmer and dryer climate predicted for the region would likely eliminate many wetland habitats thus reducing wetland density and increasing isolation. Such changes would reduce the amount of available breeding habitat and decrease gene flow and the likelihood of recolonization of local populations.

A variety of birds and mammals use riparian areas as habitat, and several species and selected guilds have been shown to respond to degradation of these ecosystems (Crronquist and Brooks 1991, 1993, Brooks et al. 1998). Louisiana waterthrush, one of the few obligate avian species in tributary watersheds of the ERMN, could serve as an integrative indicator of condition because of their dependence on interior forest habitat and clean, headwater streams (Prosser and Brooks 1998, O'Connell et al. 2003). Beaver activities frequently alter the entire structure and function of headwater streams and wetlands, and thus, their populations need to be monitored so that these effects can be assessed. Other aquatic mammals, such as mink, otter, are sensitive to bioaccumulation of contaminants found in aquatic habitats. Their availability through legal fur-trapping activities or road kills may provide a source of tissues for analyses of these contaminants.

The influence of pollutants on the biota of streams has been well documented, and forms the basis of many federal and state water quality regulations (e.g., Karr 1999, Karr and Chu 1999). The delivery pathways of pollutants can occur through atmospheric deposition, point and non-point hydrologic discharges, and movement of bioaccumulating species. Effects vary greatly, but pollutants can have immediate lethal effects (e.g., single or episodic discharges), more subtle sub-lethal impacts (e.g., changes in individual health or fecundity), or alter habitats causing harm without direct physiological consequences (e.g., loss of diverse stream substrate, absence of natural tree cavities). In addition, the strong influence of the surrounding landscape on a wetland's or stream's ability to perform a biologically-related function has become increasingly evident (e.g., Gibbs 1993, Wardrop and Brooks 1998, O'Connell et al. 2000, Strayer et al. 2003). Connectivity among aquatic habitats has been shown to affect both faunal (e.g., Gibbs 1993) and floral communities. For example, movements of vulnerable species can be hindered by dams, dikes, and culverts (e.g., bog turtles) and discontinuities among requisite habitats can affect reproductive success and genetic diversity.

How humans interact with a landscape within the physical constraints of climate and geology defines land use. Land use can be considered a major driver of the characteristics and conditions of tributary watersheds. Stream biological integrity was strongly correlated with the extent of agriculture, wetlands, and forests in the surrounding landscape (Roth et al. 1996), It is not only the type of land use that affects these watersheds, however, but also the patterns formed by the mosaic of land uses imposed over time. Of particular importance to aquatic ecosystems are the patterns that arise along riparian corridors (Jordan et al. 1993, Castelle et al. 1994, Sweeney et al. 2004). In the Mid-Atlantic region, stream reaches with wider forested riparian corridors supported higher abundance of macroinvertebrates, and processed more carbon, nitrogen, and pesticides than narrower reaches. Because of this relationship, attributes of both landscape patterns and riparian corridors can be used to assess condition (King et al. 2005).

DEVIATIONS FROM REFERENCE CONDITION CAUSED BY STRESSORS

When considering how various stressors influence tributary watersheds, it is instructive to consider deviations from reference standard conditions that support the highest levels of biological integrity. In the eastern U.S., the best attainable conditions for tributary watersheds are derived from a landscape dominated by mature forests, which produce characteristic inputs of organic matter, shade over wetlands and narrow stream corridors, and habitat for an expected set of species. In floodplains, microtopographic heterogeneity arises from the interplay of hydrologic forces, vegetative structure, and underlying soil characteristics. The resultant mosaic of wet and dry patches found in natural floodplains and along the interfaces between aquatic and terrestrial systems support a diversity of biological communities adapted to wetting and drying cycles. These physical and biological complexities interact with and upon the materials present through biogeochemical processes to produce the ecological functions and services recognized from these systems.

As humans transform the landscape, forest cover is generally reduced, replaced by agricultural, suburban, and urban land uses linked through transportation and utility corridors. The spatial extent and pattern of these changes determines the degree of alteration and degradation observed in tributary watersheds. Additionally, point sources of urban stormwater, agricultural runoff, and other pollutants can severely degrade tributary watersheds. Degrees of change can be detected through monitoring if selected attributes are used as vital signs.

RECOMMENDED VITAL SIGNS FOR TRIBUTARY WATERSHEDS

Background

In the Mid-Atlantic Region, and in particular the Appalachian Mountains, a number of initiatives have developed ecological indicators applicable to monitoring the condition of tributary watersheds. The Environmental Monitoring and Assessment Program (EMAP) of the U.S. Environmental Protection Agency developed and tested a variety of condition and stressor indicators for wadeable streams (Bryce et al. 1999, EPA 2000, Herlihy et al. 2000, Klemm et al. 2003). The Penn State Cooperative Wetlands Center (CWC) has produced a set of indicators for wetlands in the region based on rapid assessment techniques (Brooks et al. 2004) and the development of hydrogeomorphic (HGM) functional assessment models and indices of biological integrity (IBI) that were summarized by Brooks (2004). The CWC has developed and is applying a set of monitoring tools to detect changes in condition and to diagnose the relevant stressors of these valuable aquatic ecosystems. A common thread through all of these techniques is treating tributary watersheds holistically, rather than as a set of separate components. As a matter of efficiency, the first level of monitoring (Level 1) uses landscape analysis as a coarse filter to prioritize which watersheds are in most need of protective or restorative measures. With Level 1, the extent and pattern of land use can be identified as stressors. Once a watershed is selected for further study, rapid assessment methods (Level 2) are applied to refine the condition assessment and identify dominant stressors. More intensive methods (Level 3) are used to target specific sites to determine the extent of impact by stressors and to assess the integrity of biological communities (e.g., macroinvertebrates, amphibians, fish, plants) (Brooks et al. 2004). Use of this type of multi-level approach can effectively and efficiently be used to monitor and assess the ecological integrity of tributary watersheds within NPS units throughout the region.

Working toward a goal of integration of waters, the Atlantic Slope Consortium (ASC) has developed and tested an expanded rapid assessment protocol that simultaneously samples the stream, wetland, and riparian components at sites that are compiled on a watershed basis (SWR, R. Brooks et al., unpublished). Specifically, the ASC's SWR Protocol uses a stressor checklist to simultaneously record the presence of stressors to in-stream, wetland, and riparian corridor portions. When more detailed measures of condition are needed to assess either the degree of degradation or the success of restoration, HGM functional assessment models for relevant wetland subclasses in the region can be applied. IBIs have been developed for both wetland and stream macroinvertebrates, wetland and stream amphibian communities, and wetland vegetation. Fish IBIs, when appropriate for larger streams and rivers, are available (Barbour et al. 1999, EPA 2000).

Vital Signs Recommended for Tributary Watershed Monitoring in the ERMN

Based on this conceptual model of the ecology of tributary watersheds and the threats to their ecological integrity (Figures 3 and 4), and the list of candidate vital signs developed by the NPS (see the ERMN Prioritization Process), we recommend the following list of indicators to be used as vital signs for the ERMN units. The emphasis is on biological indicators which integrate across spatial and temporal scales. More detailed summary narratives of each recommended vital signs follow.

<u>Vital Sign (VS) Number – Vital Sign Name</u>
VS02 – Wet and dry deposition
VS04 - Weather and climate
VS07 - Stream / river channel characteristics
VS13 - Surface water hydrology – streams
VS14 – Water hydrology - wetlands
VS16 - Water quality – core parameters
VS18 - Invasive plants, animals, diseases – status & trends
VS19 - Invasive plants, animals, diseases – early detection
VS28 - Riparian plant communities
VS29 - Riparian birds
VS30 - Riparian mammals (may provide bioaccumulation contaminants)
VS39 - Aquatic macroinvertebrates – water quality suite
VS42 - Aquatic periphyton – algae, diatoms, fungi, bacteria, and protozoa
VS44 - Fish communities – streams; contaminants and recreation
VS46 - Vernal pond amphibians
VS47 - Streamside salamanders
VS57 - Land cover / land use change
VS58 - Landscape pattern
VS99 – Indicator taxa (new vital sign proposed)

VITAL SIGN NARRATIVES

Level 1 ► Air and Climate
Level 2 ►Air Quality
Level 3 ► Wet and Dry Deposition (VS2)

***Brief Description*: "**Deposition" refers to the deposition of, and trends in, pollutants that are carried in ambient air and deposited on National Park Service lands in the Eastern Rivers and Mountains Network (ERMN). Atmospheric deposition is the process by which airborne particles and gases are deposited to the earth's surface either through wet deposition (rain or snow), occult deposition (cloud or fog), or as a result of complex atmospheric processes such as settling, impaction, and adsorption, known as dry deposition. Although it is important to know total deposition, (i.e., the sum of wet, occult, and dry deposition) to park ecosystems, often only the wet deposition component is known, as it is the only one that is monitored routinely and extensively across the U.S. through the National Atmospheric Deposition Program (NADP). Acids, nutrients, and toxics are the primary compounds within deposition that are of concern in park ecosystems. For the most part, atmospheric pollutants are **primary predisposing and inciting factors** affecting ecosystem health.

***Significance/Justification*:** All of the ERMN parks occur within or downwind of areas of the central and eastern United States that have a significant influence from industrialization and power generation. Vehicular burning of fossil fuels in the densely populated region also contributes much to the atmospheric pollution load. These pollutants have potentially sweeping effects on the entire ERMN (Lovett 1994). Deposition effects are manifested in a variety of ways, depending on the pollutant. Direct effects include foliar necrosis and dieback in plants. In other cases, pollutants may be directly toxic to plants, animals or microorganisms. However, indirect effects that result, for example, from soil acidification and its effect on mineral cycling may be more significant in the long term. Atmospheric pollutants potentially affect resources such as water and mineral nutrients. Aquatic ecosystems, particularly in headwater areas with low buffering capacity, can become episodically acidified, resulting in significant degradation of aquatic communities. The long-term effects, such as altered litter decomposition, micro-flora and fauna, altered nutrient cycling, and acidification of aquatic ecosystems pose major threats to the health, fecundity and sustainability of the terrestrial and aquatic ecosystems and lead to an overall loss of species diversity.

Proposed Metrics: Due to the relative lack of regional data on dry and occult deposition, the ERMN will use wet deposition data reported as kilograms per hectare per year (kg/ha/yr).

***Prospective Method(s) and Frequency of Measurement*:** The ERMN will rely on wet deposition data measured at NADP sites in and near network parks. NADP measures a comprehensive suite of anions and cations; deposition rates of total wet sulfur (S) and total wet inorganic nitrogen (N) (ammonium plus nitrate ions) are included in the summaries.

***Limitations of Data and Monitoring*:** Ideally, the ERMN would evaluate total deposition, i.e., wet plus dry plus occult, to assess the threat to resources. Realistically, only wet deposition data are available. Wet deposition values will be based on interpolated data for most ERMN parks since only one park has an on-site NADP monitor. Because of meteorology and intervening terrain, interpolated deposition values may be somewhat different than those that would be based on on-site data. Wet deposition data should be compared to the results of water quality monitoring data to understand linkages between contributing areas and aquatic ecosystems. Atmospheric pollution is often a problem of regional, even global proportions, therefore it may be difficult or impossible to mitigate. Moreover, the sources of pollution are outside the parks and, therefore, cannot be controlled by the NPS.

***Key References*:**

Asman, W.A.H., Sutton, M.A., & Schjorring. 1998. Ammonia: emission, atmospheric transport and deposition. New Phytologist, 139: 27-48.

Cappellato, R., Peters, N.E., & Meyers, T.P. 1998. Above-ground sulfur cycling in adjacent coniferous and deciduous forests and watershed sulfur retention in the Georgia Piedmont, U.S.A. Water, Air, and Soil Pollution, 103: 151-171.

DeHaynes, D.H., Schaberg, P.G., Hawley, G.J., & Stimbeck, G.R. 1999. Acid rain impacts calcium nutrition and forest health. BioScience. 49: 789-800.

Driscoll, C.T., Lawrence, G.B, Bulger, A.J., Butler, T.J., Cronan, C.S., Eager, C., Lambert, K.F., Likens, G.E., Stoddard, J.L., & Weathers, K.C. 2001. Acidic deposition in the northeastern United States: sources and inputs, ecosystem effects, and management strategies. BioScience, 51(3): 180-198.

Driscoll, C.T., Driscoll, K.M., Mitchell, M.J., & Dudley, J.R. 2003. Effects of acidic deposition on forest and aquatic ecosystems in New York State. Environmental Pollution, 123: 327-336.

Fenn, M.E., Hauber, R., Tonnesen, G.S., Baron, J.S., Grossman-Clarke, S., Hope, D., Jaffe, D.A., Copeland, S., Geiser, L., Rueth, H.M., & Sickman, J.O. 2003. Nitrogen emissions, deposition, and monitoring in the western United States. BioScience, 53(4): 1-13.

Garner, J.H.B., Pagano, T., & Cowling, E.B. 1989. Evaluation of the role of ozone, acid deposition, and other airborne pollutants in the forests of eastern North America. USDA Forest Service, SE-59, pp.189.

Johnson, D.W. & Fernandez, I.J. 1992. Soil Mediated Effects of Atmospheric Deposition on Eastern U.S. Spruce-Fir Forests. In C. Eager, M.B. Adams, eds., Ecology and Decline of Red Spruce in the Eastern United States. New York: Springer-Verlag.

Leith, I.D., Murray, M.B., Sheppard, L.J., Cape, J.N., Deans, J.D., Smith, R.I., & Fowler, D. 1989. Visible foliar injury of red spruce seedlings subjected to simulated acid mist. New Phytologist, 113(3): 313-320.

Lovett, G.M. 1994. Atmospheric deposition of nutrients and pollutants in North America: an ecological perspective. Ecological Applications, 4(4): 629-650.

McLaughlin, S. B., Anderson, C. P., Hanson, P. J., Tjoelker, M. G., & Roy, W.K. 1991. Increased dark respiration and calcium deficiency of red spruce in relation to acid deposition at high-elevation southern Appalachian Mountain sites. Can. J. For. Res. 21: 1234-1244.

Shortle, W. C., K. T. Smith, R. Mincocha, G. B. Lawrence, & M. B. David. 1997. Acid deposition, cation mobilization, and stress in healthy red spruce trees. J. Environ. Qual. 26: 871-876.

McLaughlin, D. 1998. A decade of forest tree monitoring in Canada: evidence of air pollution effects. Environmental Review, 6(3-4): 151-171.

Potter, C.S., Ragsdale, H.L., & Swank, W.T. 1991. Atmospheric deposition and foliar leaching in a regenerating southern Appalachian forest canopy. Journal of Ecology, 79:97-115.

Vann, D. R., A. H. Johnson, and B. B. Casper. 1994. Effects of elevated temperatures on carbon dioxide exchange in Picea rubens. Tree Physiology. 14: 1339-1349.

Weathers, K.C., Cadenasso, M.L., & Pickett, S.T.A. 2001. Forest edges as nutrient and pollutant concentrators: potential synergisms between fragmentation, forest canopies, and the atmosphere. Conservation Biology, 15(6): 1506-1514.

***Related Environmental Issues and Linked Vital Signs*:** Atmospheric pollutants directly affect a number of ecosystem processes. In particular, soils can absorb and accumulate pollutants, altering **nutrient cycling**. Acidified soils have lower base saturation and therefore lower fertility resulting in reduced bio-productivity. Runoff, throughfall and direct input to streams and lakes can result in impacts to aquatic systems as well as to terrestrial systems which can lead to loss of sensitive species.

***Overall Assessment*:** Atmospheric deposition of sulfur and nitrogen compounds is prevalent in the EMRN region and can affect numerous ecosystem processes, including **nutrient cycling,** litter dynamics and **regeneration**. Indirect effects of pollutants may be the enabling of invasive species and the loss of T&E species due to habitat alteration or direct toxicity. Amphibian species appear to be especially sensitive to water-borne pollutants. The ERMN can rely on the existing network of NADP monitors for wet deposition data, but because the NPS cannot control sources of pollution outside park boundaries, mitigation and reclamation of damaged ecosystems will be difficult.

Level 1 ► Air and Climate
Level 2 ►Weather and Climate
Level 3 ► Weather and Climate (VS4 - tributary)

Brief Description**:** Weather and climate have the potential to affect the distribution of all species. The present geographical distribution of species can be presumed to be a consequence of past redistributions as weather and climate have changed over time leading up to the present. However, species redistributions have been shown to occur at different rates as exemplified by different rates of latitudinal movement of tree species distributions following the last glacial maximum in North America (Davis 1987). This argues for regarding species movements during climate change individually, and predicting these movements based on the ecological tolerances of each species. In contrast, all species are constrained to some extent by the ecological relationships with other species. Species with commensal, predator-prey, or other "symbiotic" interrelationships are likely to have coordinated redistributions, and thus not follow individualistic patterns. Any models taking into account the effects of climate change must recognize this duality. Species at the northern or southern limits of distributions are the ones that could serve as indicators of response to climate change. A 1.5 – 4.5 degrees C warming by the end of the twenty-first century, as indicated by Overpeck et al. (1991), could lead to as shift of southern species to the north (Solomon and Kirilenko, 1997). Species isolated geographically to the highest altitudes, such as red spruce in the southern Appalachians, could be extirpated locally (Adams et al. 1985).

Significance/Justification**:** Weather and climate are but one set of factors representing the multidimensional niche of species, and thus their current distributions. The geographical redistribution of species may have cascading effects on other dependent species. To the extent that some tributary watersheds occur at the highest altitudes, high altitude distributions are expected to be the most vulnerable. Likewise, organisms at the fringes of climatically restricted population distributions are the most vulnerable to additional stressors caused by human activities. Such species may serve as indicators of these interactions (De Groot et al.1995). As described elsewhere, Mahon (2004) provides lists of plants, vertebrates and communities of special concern in the New River Gorge, some of which may be among the first to respond to climate change.

Proposed Metrics: Prospective Method(s) and Frequency of Measurement**:** Two groups of species deserve monitoring: those that have rather distinct north and south boundaries and those that are restricted to high altitudes. For the former, population abundances and other indices of population vitality can be measured at the boundaries of the species distributions. For the latter, a similar approach may be taken for altitude. As with any measurement, and especially climatically related indicators, interannual variation can be a critical component in interpreting the relevance of long-term data to species distribution. For example, weather extremes of precipitation, temperature, storminess, daytime vs. nighttime averages, etc. may each have influences on populations locally and over short time intervals.

Limitations of Data and Monitoring: Most species of interest will be ones distributed outside of boundaries of NPS control. Consequently, any monitoring program must be driven by the distribution of the chosen indicator species rather than only the distribution in lands under federal jurisdiction of the NPS.

Key References:

Adams, H.S., Stephenson, S.L., Blasing, T.J., & Duvick, D.N. 1985. Growth-trend declines of spruce and fir in mid-Appalachian subalpine forests. Environmental and Experimental Botany, 25(4): 315-325.

Davis, M.B. 1987. Invasion of forest communities during the Holocene: Beech and hemlock in the Great Lakes Region. Page 373-393 in A.J. Gray, M.J. Crawley, and P.J Edwards (editors). Colonization, Succession, and Stability. Blackwell Publishers, Oxford, UK.

De Groot, R.S., Ketner, P., & Ovaa, A.H. 1995. Selection and use of bio-indicators to assess the possible effects of climate change in Europe. Journal of Biogeography, 22: 935-943.

Hicks, R. R., Jr. 1998. Ecology and Management of Central Hardwood Forests. John Wiley and Sons, New York. 412 pp.

Mahon, C. G. 2004. A Natural Resource Assessment for the New River Gorge National River. US Department of Interior, NPS Technical Report NPS/NER/NRTR—2004/002. 129 pp.

Overpeck, J.T., Barlein, P.J., & Webb III, T. 1991. Potential magnitude of future vegetation change in eastern North America: comparisons with the past. Science, 254(5032): 692-695.

Solomon, A.M. & Kirilenko, A.P. 1997. Climate change and terrestrial biomass: What if trees migrate? Global Ecology and Biogeography Letters, 6(2): 139-148.

Related Environmental Issues and Linked Vital Signs: Weather and climate directly affect a number of other ecosystem attributes, especially related to sensitive and T&E species, biodiversity, etc. Because climate does not act in a vacuum, other vital signs such as levels of atmospheric pollution (VS1, VS2, VS3) may interact with climate to affect organisms. Indirect effects may occur such as the enabling of invasive species and loss of focal species or communities.

Overall Assessment: Climate plays a fundamental role in terrestrial ecosystems, and particularly tributary watersheds at high altitudes. Therefore climatic changes, and associated alterations in weather patterns, have the potential to change the distribution of species and associated communities. Current models of climate change are notoriously general and geographically imprecise. In some ways, changes in species distribution may more effectively indicate climate change than the use of climate change projections to predict future distributions of species. Regardless, a great deal of uncertainty will accompany either approach, and it is the accumulation of multiple trends of many indicators that will ultimately be the most compelling evidence for change.

Level 1 ► Geology and Soils
Level 2 ►Geomorphology
Level 3 ► Stream/River Channel Characteristics (VS07)

Brief Description**:** "Stream channel characteristics" (SCC) refers to the physical component of stream habitat and includes information on stream size, sinuosity, bed roughness, channel slope, bank condition, water depths, water velocities, substratum, and the amount and type of organic matter and instream vegetation. Usually, measurements of physical habitat are collected in conjunction with hydrology (flow), water chemistry, and riparian vegetation sampling. Individual SCC variables are typically summarized using traditional statistical measures of magnitude and variance such as means and standard deviations, as well as with more complex, integrated, measures such as habitat complexity and substrate stability.

Significance/Justification: Measures of habitat quality are essential components of any long-term stream monitoring program. Along with water quality, the physical characteristics of stream channels are the main proximate determinants of biotic integrity in streams. Individual species or life stages of a single species vary with respect to physical habitat requirements and preferences, and SCC variables summarized at various spatial scales represent a multidimensional representation of individual habitat patches important to component species. Over long stream reaches, it is the diversity and stability of available habitat patches as determined by SCC, as well as the spatial and temporal relationships among them, that shape biological communities in streams (Townsend 1989, Poole 2002). Moreover, SCC indirectly affect biological communities through their influence on energy flow. Specifically, SCC variables such as bed roughness, pool-riffle ratios, and the amount of coarse woody debris within the channel are primary determinants of carbon and nutrient flow through, and retention within, lotic systems (Brookshire and Dwire 2003).

In undisturbed watersheds, SCC are determined by interactions between climate, basin size, geology, and terrain (Gordon et al. 1992). However, both natural and human induced disturbances can have profound effects on SCC and consequently biological integrity. For example, increases in the amount of impervious surfaces associated with urbanization within a watershed have been shown to cause higher storm flows which leads to bank erosion (i.e., changes in stream size), increased sedimentation (reduced substrate size and increased substrate embeddedness, especially in riffle areas), shallower and less complex pool habitats, and ultimately reduced biotic integrity (Richards et al. 1996, Snyder et al. 2003). Consequently, understanding status and trends in biological integrity of stream ecosystems requires basic information on SCC.

Proposed Metrics**:** Important SCC metrics include mean channel width, substratum size distributions (especially in riffles), substrate embeddedness, amount and size distribution of large woody debris, proportion of stream channel area with submerged and emergent vegetation, pool-riffle ratios, number and size of dispersal barriers (beaver dams, waterfalls, man-made dams and dikes), measures of bank stability, and variation in depth and flow patterns. In addition, integrated measures such as instream habitat diversity, fish cover, and substrate stability are also recommended.

***Prospective Method(s) and Frequency of Measurement*:** Two types of methods are typically used to assess stream habitat: quantitative assessments that involve detailed measurements of stream channel and bank characteristics (e.g., Rosen 1994); and visual-based rapid assessments that involve relative rankings of important stream habitat features. Quantitative assessments have the advantage of providing accurate and unbiased data that can be collected by trained field technicians. However, these measurements are time consuming and require a significant amount of field equipment. In contrast, with visual-based rapid approaches, a very large amount of information can be acquired in a relatively small amount of time with little equipment. However, these visual rankings are more sensitive to investigator bias and consequently a significant amount of training and testing is required to minimize subjectivity and ensure comparability. It is usually recommended that a single biologist conducts all visual based assessments.

If possible, a combination of the two approaches should be used with quantitative methods applied less frequently (perhaps once every five years) and rapid assessments used more often (e.g., annually). The EPA has developed and tested a visual-based habitat assessment approach which is described in Barbour et al. (1999). Specific rapid protocols have been developed for both low and high gradient stream systems. Quantitative habitat assessment protocols are described by Meador et al. (1993), Rosgen (1994), and Kaufmann and Robinson (1997).

***Limitations of Data and Monitoring*:** As previously mentioned, substantial training of field crews is required to minimize subjectivity of the qualitative rankings used in the rapid habitat assessment approach. Moreover, rankings are often affected by current weather conditions. In contrast, the quantitative approach is more expensive and time consuming and requires a significant amount of equipment. However, the

***Key References*:**

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

Brookshire, E.N.J. and K.A. Dwire. 2003. Controls on patterns of coarse organic particle retention in headwater streams. Journal of the North American Benthological Society 22:17-34.

Gordon, N.D., T.A. McMahon, and B.L. Finlayson. 1992. Stream Hydrology: an Introduction for Ecologists. John Wiley and Sons. New York. 526 pp.

Kaughman, P.R. and E.G. Robinson. 1997. Physical habitat assessment. Pages 6-1 to 6-38 in D. J. Klemm and J. M. Lazorchak (editors). Environmental Monitoring and Assessment Program. 1997 Pilot Field Operations Manual for Streams. EPA/620/R-94/004. Environmental Monitoring Systems Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Meador, M.R., C.R. Hupp, T.F. Cuffney, and M.E. Gurtz. 1993. Methods for characterizing stream habitat as part of the national water-quality assessment program. U.S. Geological Survey Open-File Report, Raleigh, North Carolina. USGS/OFR 93-408.

Poole G.C. 2002. Fluvial landscape ecology: addressing uniqueness within the river discontinuum. Freshwater Biology 47:641-660.

Richards, C., L.B. Johnson, and G.E. Host. 1996. Landscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Sciences 53:295-311.

Rosgen, D.L. 1994. A classification of natural rivers. Catena 22:169-199.

Snyder, C.D., J.A. Young, R. Villella, and D.P. Lemarie. 2003. Influences of upland and riparian land use patterns on stream biotic integrity. Landscape Ecology 18:647-664.

Townsend C.R. 1989. The patch dynamics concept of stream community ecology. Journal of the North American Benthological Society 8:36-50.

***Related Environmental Issues and Linked Vital Signs*:** SCC are primary determinants of stream biotic assemblages and consequently should be considered along with any stream faunal groups selected for monitoring (VS39, VS41, VS42, VS43, VS44, and VS47). In turn, SCC are themselves effected by weather (VS4), geology (VS11) and riparian (VS28) and upland (VS57) land cover characteristics.

***Overall Assessment*:** SCC should be considered a high priority vital sign because they are both drivers of ecological integrity in streams, and sensitive to many of the sources of environmental degradation noted for ERMN.

Level 1 ► Water
Level 2 ►Hydrology
Level 3 ► Surface Water Hydrology – Streams, Rivers, Wetlands (VS13,14)

***Brief Description*:** Tributary watersheds consist of a complex of streams, riparian zones, and wetlands that are supported by various combinations of precipitation, surface water, and groundwater. The physiographic origins, flow patterns, hydrodynamics, and water quality attributes determine the mosaic of aquatic habitats in these systems. Understanding hydrologic reference conditions is critical for diagnosing hydrologic stressors.

***Significance/Justification*:** Understanding and measuring the hydrology of tributary watersheds is central to many other ecological aspects of assessing the condition of these systems (Forman 1995, Thorp et al. In press). As the primary driver of these systems (Mitsch and Gosselink 2000), and as a link between climate and weather indicators, hydrology should be considered a core vital sign. From an energy perspective, the ecological integrity of stream communities in mid-reach streams is determined mostly by factors that affect retention, transport, and the quality of organic matter from headwater areas upstream, and by factors that influence instream primary production within mid-reach areas. The effects of non-point source pollutants associated with agriculture and urban land use in upstream or adjacent landscapes (both of which are significant concerns in ERMN) have been shown to affect energy pathways in these reaches. Herbicides and increased sediment inputs have been shown to reduce overall instream primary production with subsequent changes in macroinvertebrate diversity and production (Guasch et al. 1998). Acidification of stream habitats has also been shown to alter primary production in streams.

***Proposed Metrics*:** Hydrologic measurements are relatively standardized. The placement of equipment to acquire those data, however, must be strategically considered. Each park unit should determine its needs for hydrologic data, potential partnering opportunities, and costs in order to design an appropriate hydrologic monitoring system. Given the importance of hydrology to these systems, it is important to capture as much of these data as possible. In some case, particularly for floodplains and wetlands, observed hydrologic indicators can be used as surrogates to quantitative measures.

***Prospective Method(s) and Frequency of Measurement*:** Hydrologic data for streams typically originates from gaging stations, from which flow rates, frequency of flooding, and other hydrologic measurements can be derived. Gaging stations, however, can only be installed on limited reaches due to relatively high expense of the equipment. Modeling and other types of simulations can be used to extend empirical measurements across other streams. The network of gaging stations can be extended through partnering efforts with other agencies and organizations. Groundwater measurements for streams and wetlands typically are taken from wells and piezometers placed at various depths into soil and geologic strata. Automated or hand measurements can be taken. The sampling regime for hydrologic measurements should be coordinated with water quality data collection to allow the computation of loadings and to increase efficiency.

***Limitations of Data and Monitoring*:** Due to the expense of installing and maintaining an extensive hydrologic monitoring system, careful consideration should be given the locations of sampling stations and types of surface water and groundwater data.

***Key References*:**

Forman, R. T. 1995. Land mosaics: the ecology of landscapes and regions. Cambridge University Press. 632pp.

Guasch, H., N. Ivorra, V. Lehmann, M. Paulsson, M. Real, and S. Sabater. 1998. Community composition and sensitivity of periphyton to atrazine in waters: the role of environmental factors. Journal of Applied Phycology 10:203-213.

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands, 3rd edition. John Wiley & Sons, Inc., New York, NY.

Thorp, J. H., M. C. Thoms, and M. D. Delong. (in press) A model of biocomplexity in river networks across space and time. journal?

***Related Environmental Issues and Linked Vital Signs*:** Hydrologic measurements are key to diagnosing stressors in tributary watersheds. Whether obtained as quantitative measurements or as qualitative observations, documenting deviations from reference hydrologic conditions is important.

***Overall Assessment*:** Hydrologic data are neither easy nor inexpensive to obtain, but their utility to monitoring the condition of aquatic ecosystems is critical.

Level 1 ► Water
Level 2 ►Water Quality
Level 3 ► Core Parameters (VS16, tributary)

***Brief Description*:** Water quality in its broadest interpretation refers to all of the factors that influence the hydrology, biogeochemistry, and habitats of aquatic organisms. In a more typical interpretation, water quality of surface water relates to physical and chemical factors such as temperature, pH, conductivity, nutrient content, sediment load, and toxicant presence. For ground water, sediment load could be eliminated as a factor. Sources of degraded water quality can originate from point sources and non-point sources. Point sources are represented primarily by wastewater treatment systems from municipal and industrial sources. Non-point sources are more diffuse, and are influence by land uses. With advances and implementation of wastewater treatment over the past three decades, many point sources have been reduced greatly, although problems have not been eliminated totally. Degraded water quality from non-point sources continues to be a chronic and difficult problem to resolve. Land use normally is the major factor affecting water quality. Rowcrop agriculture and various urban land uses have strong effects of water quality in addition to direct discharges from point sources (Lowrance et al. 1984). Headwater streams are strongly influenced nonpoint sources because they are the first to receive surface water it passes from land-based activities to low order streams (Brinson 1993). Unless in-stream processes improve water quality, nutrients and other contaminants will be transferred to higher order streams downstream (Bayley 1995, Jones and Mulholland 2000). Natural variation in water hardness, acidity, and dissolved oxygen are major controls over species composition of aquatic communities. These controls, however, may be easily overwhelmed by eutrophication, excessive sediments, and toxicants that stress aquatic organisms and eliminate whole suites of species. In tributary watersheds, water chemistry is more reflective of the geologic and topographic characteristics of the landscape than for larger rivers. The complex geology of the Appalachians can create circumstances where relatively short stream reaches and individual tributaries can have different water chemistry than their neighbors (USEPA 2000, Snyder et al. In review.) Such variability produces extraordinary biodiversity.

Significance/Justification: Eutrophication from excess nutrients (e.g., nitrogen and phosphorus) can be a significant stressor in tributary watersheds. Over time, eutrophication typically alters energy pathways by increasing primary production, which often results in lower dissolved oxygen concentrations resulting from oxygen demand from accumulated organic matter. These changes usually lead to highly productive, but taxonomically and trophically simple biological communities in both streams and wetlands (Sandin and Johnson 2000). Herbicides also disrupt energy pathways, but they cause reductions in instream primary production, and pesticides directly affect survival and reproduction of populations of invertebrates and fish. Excess turbidity caused by high levels of suspended sediment decreases oxygen levels and photosynthesis rates, impairs the respiration and feeding of aquatic organisms, destroys fish habitat, and kills benthic organisms (Johnston 1993). In wetlands, high sedimentation rates decrease the germination of many wetland plant species by eliminating light penetration to seeds, lowering plant productivity by creating stressful conditions, and slowing decomposition rates by burying plant material (Jurik et al. 1994, Vargo et al. 1998, Wardrop and Brooks 1998).

***Proposed Metrics*:** Protocols for monitoring the status and changes in water quality are well established and have been used for decades. As a practical matter, the purpose and scope of monitoring should depend on the issues being addressed. For example, if water quality problems are suspected to be the result of acid mine drainage, and remediation practices are implemented, intensive sampling of acidity, heavy metals, and sensitive biota may be the indicators of choice. On the other hand, if the question revolves around protecting habitat quality, and no specific problems are apparent or known, sampling for periphyton, benthic macroinvertebrates, and fish at infrequent intervals may be the method of choice. There are a number of established principles for designing monitoring programs to detect effects of human activities (Downes et al. 2002).

***Prospective Method(s) and Frequency of Measurement*:** As for the proposed metrics, the method and frequency of measurements should match the purpose of the sampling program. Methods range from characterizing benthic, diatom, or fish communities (IBIs) (Karr 1999, Karr and Chu 1999) to analysis for specific chemical components. Problem areas may be identified with a number of spatially explicit analytical tools, such as EPA-developed Analytical Tools Interface for Landscape Assessments (ATtILA).

***Limitations of Data and Monitoring*:** Many streams have been sampled for water quality through state and federal programs. However, headwater tributaries are among the least sampled on a routine basis, simply because they are so abundant and occur everywhere in the landscape. In remote areas, sampling is difficult because of access problems, and few historical data exist as points of reference. The chemical and sediment components of streams may vary widely depending on stream discharge. Stream discharge varies seasonally and with differences in base flow vs. storm flow. Time of day in which data are recorded can have an influence on the concentration of dissolved oxygen. For these and other reasons, the intent of a water quality sampling program should be carefully evaluated before a commitment is made to dedicate resources to the time and expense necessary for program implementation. Priority for remedial action can be assigned to sites that have been identified as being impaired, such stream segments that are on 305(b) and 303(d) lists and where other assessments have identified problems. Sites should be identified that are slated to have application for National Pollution Discharge Elimination System (NPDES) permits, changes in landuse/landcover, construction of roads, and additional housing and other development activities. Sampling programs to detect factors associated with climate change require special attention to planning and review (Grimm 1993, Halpin 1997).

***Key References*:**

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols or use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, 2nd ed. EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Bayley, P. B. 1995. Understanding Large River-Floodplain Ecosystems. BioScience 45:153-158.

Brinson, M.M. 1993. Gradients in the functioning of wetlands along environmental gradients. Wetlands 13:65-74.

Downes, B.J. and others. 2002. Monitoring Ecological Impacts: Concepts and practice in flowing waters. Cambridge University Press. Cambridge, UK.

Grimm, N. B. 1993. Implications of climate change for stream communities. Pages 293-314 *in* P. M. Kareiva, J.G. Kingsolver, and R.B. Huey, editors. Biotic interactions and Global Change. Sinauer Associates, Sunderland, Massachusetts, USA.

Halpin, P. N. 1997. Global climate change and natural-area protection: management responses and research directions. Ecological Applications **7**:828-843.

Johnston, C. A. 1993. Mechanisms of wetland-water quality interaction. Pages 293-299 in G. A. Moshiri, editor. Constructed Wetlands for Water Quality Improvement. Lewis Publishers, MI.

Jones, J. B. and P. J. Mulholland. 2000. Streams and ground waters. Academic Press, San Diego, California, 425pp.

Jurik, T. W., S.-C. Wang, and A. G. v. d. Valk. 1994. Effects of sediment load on seedling emergence from wetland seed banks. Wetlands 14:159-165.

Karr, J. R. 1999. Defining and measuring river health. Freshwater Biology 41: 221-234.

Karr, J. R and E. W. Chu. 1999. Restoring Life in Running Waters: Better Biological Monitoring. Island Press, Washington, D.C. 206 pp.

Lowrance. R.R., R.L. Todd, J. Fail, O. Hendrickson, R. L3eonard, and L.E. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. BioScience 34:374-377.

Sandin, L. and R. K. Johnson. 2000. The statistical power of selected indicator metrics using macroinvertebrates for assessing acidification and eutrophication of running waters. Hydrobiologia 422:233-243.

Snyder, C. D., J. A. Young, and B. M. Stout, III. Landscape perspective on aquatic habitats in Canaan Valley, West Virginia. Northeastern Naturalist, (in review).

U.S. Environmental Protection Agency. 2000. Mid-Atlantic Highlands Streams Assessment. EPA/903/R-00/015. U.D. Environmental Protection Agency Region 3. Philadelphia, PA. 64pp.

Vargo, S. M., R. K. Neely, and S. M. Kirkwood. 1998. Emergent plant decomposition and sedimentation: response to sediments varying in texture, phosphorus content and frequency of deposition. Environmental and Experimental Botany 40:43-58.

Wardrop, D. H., and R. P. Brooks. 1998. The occurrence and impact of sedimentation in central Pennsylvania wetlands. Environmental Monitoring and Assessment 51:119-130.

Related Environmental Issues and Linked Vital Signs**:** Water quality potentially responds to changes in land cover/land use (VS 57, VS 58), in atmospheric and climatic patterns (VS1-VS4), geology and soils (VS6- VS12), and hydrologic features (VS14, VS15). Because human activities often are involved in the introduction of invasive species, and human habitation is relevant. Visitor usage (VS54) can locally affect aquatic ecosystem through changes in productivity and nutrient dynamics (VS59, VS61). In short, nearly all activities that occur in watersheds have the potential to alter the pattern that exists on the landscape is a reflection of the sum of the abiotic, biotic and anthropogenic factors that interact over it.

Overall Assessment**:** Given all of the related environmental links and issues listed above, water quality in its various dimensions should become a critical component of any vital signs program. ERMN parks are themselves part of a larger landscape, are affected by actions that take place beyond their boundaries, and aquatic ecosystems can be one of the pathways that provide a conduit for transporting problems through park boundaries. The relatively long history of and experience in water quality monitoring can provide a suite of effective tools to enable managers to anticipate changes and take remedial actions, when necessary.

Level 1 ► Biological Integrity
Level 2 ►Invasive Species
Level 3 ► Invasive Plants, Animals, Diseases – Status and Trends (VS18 – emphasis on aquatic species)

***Brief Description*:** "Invasive plants, animals, diseases – status and trends" is a very broad subject, including 1) invasive plants and animals whose primary effect is displacement of native species and 2) species of exotic insects, animals or pathogens that attack and cause injury or death to native species. Examples include purple loosestrife, zebra mussels, and exotic crayfish. An abundance of invasive plants and animals is often associated with disturbed or degraded ecosystems (Dachler and Carino 2000); therefore their presence serves as an **indicator** of ecosystem health. On the other hand, invasive species, including insect and disease pests, can dramatically alter an ecosystem (serving as an **inciting factor** for ecosystem decline), thus directly affecting processes such as **succession, nutrient cycling, and food webs**. Furthermore, the altered ecosystem state may result in a system that is **unhealthy,** has lower **diversity** and having reduced **fecundity** of native species. Invasive species, including insects and diseases, have resulted in dramatic historic changes to numerous ecosystems in North America, including the ERMN area. The recent invasion of the hemlock wooly adelgid is an interesting example of a terrestrial pest that could significantly impact tributary watersheds (e.g., McClure and Cheah 1999).

Significance/Justification: Native plants and animals, that make up a particular ecosystem have co-evolved over millions of years, therefore native ecosystems have developed a state of dynamic equilibrium. The introduction of non-native species into a system can upset this balance. Because of the globalization of human activities, including travel, shipping and deliberate species introduction for food and agricultural purposes, many species have been moved from their native ranges and have been introduced to exotic environments around the world. In most cases, these species have been unsuccessful or have blended into the local environment with minor impacts. But for some species, their introduction has led to their becoming "invasive". This term refers to the condition that exists when a non-native plant or animal becomes highly aggressive in its new environment and causes habitat destruction, replacement of native species or results in damaging outbreaks (e.g., Davis et al. 2000). National parks are especially vulnerable to species invasion because of the large number of visitors who enter the parks and serve as potential vectors of invasive organisms. Invasive organisms can bring about alterations in **species composition, bio-productivity, and nutrient cycling**, changing the **diversity, vigor and fecundity** of the ecosystem. The direct effects of an invasion include species displacement, infestation, and mortality of host species, but indirect effects such as shifts in species composition, altered nutrient cycling, modified temperature and light regimes, and increased demand for oxygen. The introduction of organisms has resulted in greater and more lasting ecosystem damage than virtually anything brought about by humans in recent history (Pimentel et. al 2000, With 2002).

***Proposed Metrics*:** In situations where an invading organism has not yet fully colonized a suitable habitat, the metric chosen to describe the colonization is usually the rate of advancement of the infestation or killing front. In areas where infestation or invasion has already occurred, the numbers of invading organisms per unit area or the proportion of the suitable habitat that has been colonized can be a valuable metric. Finally, the presence and impact of an invasive organism or disease is often measured by the number or proportion of hosts that are colonized or killed. This can be more difficult in aquatic ecosystems than in terrestrial ones.

***Prospective Method(s) and Frequency of Measurement*:** Surveys of invasive plants and animals in aquatic ecosystems are conducted by federal and state agencies. For newly-introduced organisms that are potentially damaging, records and surveys are conducted by the USDA, Animal and Plant Health Inspection Service. Before any in-house programs are undertaken by the ERMN, this information should be investigated to determine whether or not it meets the needs of the NPS. Furthermore, hazard rating systems that have been developed, especially in the case of insects and diseases, primarily for terrestrial ecosystems, but they may be useful in determining whether or not a particular park is likely to have a problem with an invading organism. Once it is determined that a need exists for additional on-site surveys for an invading organism, the appropriate sampling scheme should be developed and tailored to the specific situation. With a problem as broad and diverse as invasive plants – animals, insects and diseases, surveys will need to be developed that are capable of detecting damaging populations and that fulfill the needs of the ERMN.

***Limitations of Data and Monitoring*:** Perhaps the greatest limitation of monitoring for invasive organisms is the sheer magnitude of the task. The ERMN parks occupy extensive areas of land and are situated in areas with extensive and remote components. Organisms can quickly spread from non-system lands onto parks. Invasive organisms can persist below detection levels and rapidly explode into outbreaks when favorable conditions occur. Data collected only on NPS lands will be of limited value in predicting the ambient population levels and therefore, may not be useful in preventing spread of organisms from adjacent ownerships. It incumbent upon the NPS to choose carefully which organisms to focus on, concentrating on those most likely to do significant damage to the parks and to utilize data collected by other agencies, whenever possible.

***Key References*:**

Daehler, C.C. & Carino, D.A. 2000. Predicting invasive plants: prospects for a general screening system based on current regional models. Biological Invasions 2: 93-102.

Davis, M.A., Grime, J.P. & Thompson K. 2000. Fluctuating resources in plant communities: a general theory of invasibility. Journal of Ecology 88: 528-534.

McClure, M.S., & Cheah, C.A.S.-J. 1999. Reshaping the ecology of invading populations of hemlock woolly adelgid, *Adelges tsugae* (Homoptera: Adelgidae), in eastern North America. Biological Invasions 1: 247-254.

Pimentel, D., Lach, L., Zuniga, R., & Morrison, D. 2000. Environmental and economic costs of nonindigenous species in the United States. BioScience 50(1): 53-65.

With, K.A. 2002. The landscape ecology of invasive spread. Conservation Biology 16(5): 1192-1203.

***Related Environmental Issues and Linked Vital Signs*:** Species invasion could be linked with Air and Climate, such that an altered climatic regime may predispose a site to being invaded. Many invasive aquatic species are spread either in the water column or by translocation from waterbody to waterbody by vectors such as birds or boats. Invasive species may displace plants and/or animals from unique natural communities, and this is especially true for T&E species, which may be living close to the limits of their existence in the absence of aggressive competitors.

***Overall Assessment*:** Invasive plants, animals, diseases – status and trends is a very broad topic, and includes both exotic invasive species that displace natural species or communities as well as insects and diseases that injure or kill native species. These agents are, however, some of the most damaging of those affecting both terrestrial and aquatic ecosystems. Their spread is directly related to human activities, either deliberate, accidental or unintentional. This makes them all the more significant in National Parks where human visitation rate is high. There may be opportunities to share monitoring costs with partners, but for certain key species, the NPS may wish to develop their own on-site survey data. The decisions regarding which species and how to sample for them should be weighed carefully, since valid surveys may be difficult, expensive and time consuming.

Level 1 ► Biological Integrity
Level 2 ► Focal Species or Communities
Level 3 ► Riparian Plant Communities (VS28, tributary)

Brief Description**:** Riparian plant communities are particularly vulnerable to invasive species because their linear nature exposes them to large areas containing potential invaders (Simberloff et al. 2005). The range of conditions of riparian zones varies widely in tributary watersheds because site conditions range from those with saturated soils to soils that are well drained and infrequently flooded. Given this range of conditions, there is little selective pressure against any particular group of species with narrow habitat preferences. Species that are dispersed by wind or water can most easily invade, and roads and trails provide additional corridors for effective dispersal. Further, disturbance factors, such as the development of point bars on rivers, opening of forest canopies by storms, and alteration of floodplains by beaver activity tend to expose sites to colonization of invasives. Once established, invasive species may compete for light, water, and nutrient resources, all of which are generally abundant in riparian areas.

Significance/Justification: Most plant species classified as invasive tend to concentrate along forest edges other areas of disturbance (Woods 1997, Walker and Smith 1997). There are few tree and shrub species, such as *Ailanthus altissima* (tree-of-heaven) and *Eleagnus angustifolia* (Russian olive), that may form monospecific stands (Miller 2004). Once established, it is difficult for native trees to compete with them. Several shrubs and vines can form dense growths in disturbed areas and forest edges, including *Ligustrum sinense* (Chinese and other privets) , *Lonicera* spp. (Japanese and other honeysuckles), *Celastrus orbiculatus* (oriental bittersweet), and *Pueraria montana* (kudzu). *Microstegium virmineum* (Japanese stiltgrass) is a grass can become particularly abundant in along stream banks and in floodplains. It is shade tolerant and a prolific seeder, and thus easily disperses. Based on its capacity to out-compete other ground covers, especially in shade, the species has the capacity to suppress other herbaceous species. *Polygonum cuspidatum* (Japanese knotweed) is becoming in increasingly prevalent. *Lythrum salicaria* (purple loosestrife) and *Phragmites communis* are obligate wetland plants found in some riparian settings (Galatowitch et al. 1999). Global warming may expand the ranges of many southern invasives into riparian areas of the ERMN. Since the rate of expansion of plant species is not predictable from ecological traits (Clark et al. 1998), empirical data are needed to follow trends in real time.

Proposed Metrics**:** Classification and inventory are the first steps in the assessment of any natural resource. If an agreed-upon list of potentially problematic species can be developed, and vulnerable sites for invasion within and surrounding each of the NPS lands are identified, this information can provide the basis for an inventory to track the occurrence and spread of invasive species. Baseline data generally are not available.

Prospective Method(s) and Frequency of Measurement**:** Annual surveys of the vulnerable sites in and around the ERMN sites would provide information on trends and conditions for riparian invasive plants. Since many invasive species tend to disperse along highways and trails, sampling sites could be located where these conduits cross stream channels. Abundance measures should be developed to characterize the areal distribution and patchiness. A recent protocol has been developed to standardize the assessment of non-native invasive species (Morse et al. 2004).

***Limitations of Data and Monitoring*:** To track changes over time, monitoring sites need to be established to illustrate where invasive species are absent as well as where they are present. Little training would be needed to recognize invasives because there are few of them, most are easily identified, and many are already familiar to most naturalists.

***Key References*:**

Clark, J. S., C. Fastie, G. Hurtt, S. T. Jackson, C. Johnson, G. A. King, M. Lewis, J. Lynch, S. Pacala, C. Prentice, E. W. Schupp, T. Webb, and P. Wyckoff. 1998. Reid's paradox of rapid plant migration. BioScience 48:13-24.

Galatowitsch, S. M., N. O. Anderson, and P. D. Ascher. 1999. Invasiveness in wetland plants in temperate North America. Wetlands 19:733-755.

Miller, J.H. 2004. Nonnative Invasive Plants of Southern Forests. USDA Forest Service. Southern Research Station, General Technical Report SRS-62, Asheville, NC

Morse, L.E., J.M. Randall, N. Benton, N. Hierbert, and S. Lu. 2004. An invasive species assessment protocol: evaluating nonnative plants for their impact on biodiversity, version 1. NatureServe, Arlington, VA. Available at http://www.natureserve.org/library/invasiveSpeciesAssessmentProtocol.pdf (accessed 25 April 2005)

Walker, L. R., and S. D. Smith. 1997. Community response to plant invasion. Pages 69-86 in J. O. Luken and J. W. Thieret, editors. Assessment and Management of Plant Invasions. Springer, New York, NY.

Woods, K. D. 1997. Community response to plant invasion. Pages 56-68 *in* J. O. Luken and J. W. Thieret, editors. Assessment and Management of Plant Invasions. Springer, New York, NY.

***Related Environmental Issues and Linked Vital Signs*:** Climate change could expand the range northward of a number of exotic species that are presently confined to warmer climates (VS 4). Because disturbance plays a large role in invasion, VS57 should be applicable.

***Overall Assessment*:** There are few examples of invasive species causing the extirpation of other plant species, except on a small, site-specific basis. Often, invasives such kudzu seem prevalent because they occupy disturbed areas along forest edges where they are conspicuous. Areas with full canopy cover are unlikely to support most invasives, with *Microstegium virmineum* an exception because of its shade tolerance. It is recommended that a modest inventory program be set up so that the spread of exotic species along riparian corridors is noted. It could be coordinated with a program to follow the phenology of plants.

Level 1 ► Biological Integrity
Level 2 ► Focal Species or Communities
Level 3 ► Riparian Birds (VS29)

Brief Description**:** Riparian birds have a demonstrated utility as integrative indicators of the condition of tributary watersheds, including stream, wetlands and riparian habitats. A variety of bird species use riparian areas as habitat, and several species and selected guilds have been shown to respond to degradation of these ecosystems (Croonquist and Brooks 1993, Brooks et al. 1998). Louisiana waterthrush (*Seirius motacilla*, LOWA), one of the few obligate avian species in tributary watersheds of the ERMN, could serve as an ideal vital sign because of their dependence on interior forest as breeding habitat and use of clean, headwater streams for foraging (Prosser and Brooks 1998, O'Connell et al. 2003). Other songbird species are considered facultative in their use of tributary habitats, but their collective use, as measured as through community composition (e.g., Index of Biological Integrity) can provide confirming information about the condition of the landscapes surrounding tributary watersheds.

Significance/Justification: Birds are well known to the public, and therefore, can garner significant public support and interest if used to express the condition of park units. Connectivity among aquatic habitats has been shown to affect faunal communities, including birds (Croonquist and Brooks 1993, Gibbs 1993). For example, movements of vulnerable species can be hindered by discontinuities among requisite habitats, which in turn can affect reproductive success and genetic diversity. Riparian songbirds are likely to respond primarily to changes in habitat structure and fragmentation, and less so to declining water quality. Waterbirds (i.e., waterfowl, shorebirds, and wading birds) are, in general, more common in larger waterbodies, such as emergent wetlands, lakes, and the floodplains and shores of large rivers. Thus, their utility as a vital sign for tiributary watersheds is limited.

Proposed Metrics**:** For non-obligate avian species, an Appalachian Bird Community Index (BCI) has been developed for the ecoregions relevant for the ERMN and for its geographic extent (O'Connell et al. 2000, O'Connell et al. (report). Scores for the overall BCI and for individual metrics can be used to assess condition of both terrestrial and tributary habitats. Density of LOWA breeding pairs may have utility (O'Connell et al. 2003). If census data are not available, habitat suitability index (HSI) models are available for several key species (e.g., Prosser and Brooks 1998), and can be easily implemented by field personnel.

Prospective Method(s) and Frequency of Measurement**:** Protocols for censusing songbirds are standardized, which increases the likelihood of acquiring high quality data. For songbirds, presence / absence data and relative abundance can be collected during the breeding season using auditory and visual identification by trained observers. As few as 1-3 observations per site are sufficient to gather useful data. Equipment needs are minimal, and personnel costs are reasonable.

Limitations of Data and Monitoring**:** There are two primary limitations to using riparian birds as vital signs; relatively narrow sampling window corresponding to accepted dates during the breeding season (approximately 8 weeks for the ERMN region) and the requirement of using observers trained in auditory and visual identifications of birds. Although the latter is essential for collecting quality data, trained birders are readily available.

***Key References*:**
Adamus – bioassmt

Brooks, R. P., T. J. O'Connell, D. H. Wardrop, and L. E. Jackson. 1998. Towards a regional index of biological integrity: the examples of forested riparian ecosystems. Environ. Monit. Assmt. 51:131-143.

Croonquist, M. J., and R. P. Brooks. 1993. Effects of habitat disturbance on bird communities in riparian corridors. J. Soil Water Conserv. 48(1):65-70.

Gibbs, J. P. 1993. Importance of small wetlands for the persistence of local populations of wetland-associated animals. Wetlands 13(1):25-31.

O'Connell, T. J., L. E. Jackson, R. P. Brooks. 2000. Bird guilds as indicators of ecological condition in the central Appalachians. Ecological Applications 10(6):1706-1721.

O'Connell, T. J., R. P. Brooks, S. E. Laubscher, R. S. Mulvihill, and T. L. Master. 2003. Using bioindicators to develop a calibrated index of regional ecological integrity for forested headwater ecosystems. Report No. 2003-01, Penn State Cooperative Wetlands Center, Final Report to U.S. Environ. Prot. Agency, STAR Grants Program, Washington, DC. 87pp.+app.

Prosser, D. J., and R. P. Brooks. 1998. A verified habitat suitability index for the Louisiana Waterthrush. J. Field Ornith. 69(2):288-298.

***Related Environmental Issues and Linked Vital Signs*:** Riparian birds can provide linkages to terrestrial ecosystems which greatly affect the condition of tributary watersheds. Brooks et al. (1998) proposed an integrative approach to assessing condition of these systems using LOWA and riparian bird communities as two of several potential metrics.

***Overall Assessment*:** Use of riparian songbirds is recommended as a suitable indicator for tributary watersheds with utility for terrestrial ecosystems as well.

Level 1 ► Biological Integrity
Level 2 ► Focal Species or Communities
Level 3 ► Riparian Mammals (VS30)

***Brief Description*:** Aquatic mammals, such as mink (*Mustela vison*) and river otter (*Lontra canadensis*), are sensitive to bioaccumulation of contaminants found in aquatic habitats. Their availability through legal fur-trapping activities or road kills may provide a source of tissues for analyses of these contaminants. In addition, park units where trapping is not permitted may provide refugia for these two riparian predators, and thus, warrant monitoring. Beaver (*Castor canadensis*) activities frequently alter the entire structure and function of headwater streams and wetlands, and thus, their populations need to be monitored so that these effects on other species, habitats, and park facilities can be assessed.

***Significance/Justification*:** Mink and river are carnivores at the top of food webs, thus, they are sensitive to accumulation of contaminants (e.g., heavy metals, PCBs) that enter aquatic ecosystems. If such contaminants are suspected or possible, and sources of tissue samples are readily available (e.g., legal fur trapping, road kills), then a modest monitoring program may justified. Also, river otter can be an attractant for visitors and recreationists, so for some units, periodic presence / absence and/or density surveys may be warranted. In selected units, species of conservation concern, such as water shrews (*Sorex palustris*) or river otters, may warrant the use of targeted protocols for actual capture or photo-capture to confirm their existence. The presence of beaver can significantly alter aquatic and vegetation features of tributary watersheds, with resultant habitat and/or economic damage. Thus, monitoring their presence and the extent of areas affected may be warranted.

***Proposed Metrics*:** For bioaccumulation studies, the concentration of suspected contaminants in target tissues (e.g., fat, reproductive organs) can be measured by qualified laboratories. Density measures derived from observed sign or captures of individuals would suffice for the other purposes.

***Prospective Method(s) and Frequency of Measurement*:** Standard methods are available for most of the suggested approaches to monitoring riparian mammals. The expected frequency of measurement is likely to be seasonal or annual. For detection of populations, inexpensive field monitoring protocols have been developed for river otters and beaver for the Delaware Water Gap unit (Swimley et al. , Serfass). Standard trapping protocols are available for detecting shrews and other small mammals.

***Limitations of Data and Monitoring*:** The primary limitation to using riparian mammals as a vital sign is the low density of the carnivores (i.e., mink, river otter, shrew), which can translate to limited data availability. Beaver, however, can be common and are readily observable.

***Key References*:**
Serfass and Brooks
Serfass
Swimley

***Related Environmental Issues and Linked Vital Signs*:** Riparian mammal species, although few in number, can be important as sentinels (mink, river otter) and agents (beaver) of environmental change.

***Overall Assessment*:** Use of riparian mammals as a vital sign is recommended in selected situations. Their use may be appropriate where park units serve as refugia for aquatic furbearers that are legally harvested outside parks, and where bioaccumulating contaminants are suspected to be present.

Level 1 ► Biological Integrity
Level 2 ► Focal Species or Communities
Level 3 ► Aquatic macroinvertebrates (VS39)

Brief Description**:** "Aquatic macroinvertebrates" refers to aquatic and semi-aquatic invertebrates that inhabit the stream bottom (i.e., benthic) and can be observed without the aid of a microscope. Most biological monitoring programs that use aquatic macroinvertebrates derive a suite of metrics from field samples that are based on the taxonomic and trophic structure and composition of the entire assemblage to infer ecological condition.

Significance/Justification**:** Aquatic macroinvertebrates are a vital component of all healthy stream ecosystems. They are instrumental in nutrient and carbon dynamics and are themselves an important link in stream food webs (Webster 1983). Moreover, unlike fish and periphyton (i.e., benthic algae), aquatic macroinvertebrate assemblages are both productive and diverse in virtually all undisturbed streams with permanent flow (Lenat et al. 1980). This is an important consideration in ERMN because many of the smaller tributary streams of component parks have steep gradients and numerous natural barriers that impede the movement of fish, as well as dense canopies that restrict light and consequently limit algal productivity. As a result, fish and periphyton assemblages are often represented by very few species even in undisturbed streams. Other advantages of using benthic macroinvertebrate assemblages to monitor streams include: 1) they are good indicators of local conditions because most benthic species are either sessile or have limited migration patterns through their aquatic phases; 2) they exhibit wide variation in tolerance among species and life stages to environmental stresses; 3) many species have long life cycles relative to other groups which allows inference regarding temporal trends; and 4) sampling aquatic macroinvertebrate assemblages is relatively easy and inexpensive, and has minimal effects on resident biota (Rosenberg and Resh, 1993, Barbour et al. 1999; and references therein). In addition, because aquatic macroinvertebrates have been by far the most commonly used group for biological monitoring of aquatic habitats in North America, a large suite of aquatic macroinvertebrate summary metrics have been evaluated with respect to natural variation and responses to numerous sources of degradation (Rosenberg and Resh 1993).

Proposed Metrics: Numerous individual assemblage response metrics can be easily calculated from macroinvertebrate sample data. However, the accuracy of measures (i.e., ability to detect impact when one occurs or the failure to detect impact when one does not occur) varies considerably among metrics and within metrics among different types of stressors. Numerous evaluations regarding the accuracy and responsiveness of many of the commonly used aquatic macroinvertebrate metrics have been done, and thorough discussions of these efforts can be found in Jackson and Resh (1993) and Barbour et al. (1999). Metrics found to be consistently robust in terms of detecting impact include several taxonomic richness measures such as total taxa richness, number of taxa of the orders Ephemeroptera, Plecoptera, and Trichoptera (i.e., EPT richness), various community similarity indices such as Margalef's Index and Simpson's Index, and some functional metrics such as the proportion of shredders (Jackson and Resh 1993). However, calculation of a much larger suite of metrics is advisable for several reasons. First, the time and expense required to calculate dozens of metrics is small relative to the time it takes to collect samples. Second, the post-hoc evaluations of the accuracy of aquatic macroinvertebrate metrics described above were designed to assess the usefulness of metrics over wide regions including different types of streams exposed to many different stressors and using information collected by different investigators using different collection methods. It is likely that within a smaller region involving a smaller set of stream types (such as ERMN) and more standardized methods and personnel, that other metrics will be revealed to be robust indicators of change. A broader list of potential macroinvertebrate metrics can be found in Resh and Jackson (1993), Kerans and Karr (1994), and Barbour et al. (1999). Various integrated measures that combine scores generated for multiple metrics into a single score have also been developed for various types of streams and geographic regions (e.g., Kerans and Karr 1994, DeShon 1995).

***Prospective Method(s) and Frequency of Measurement*:** The US Environmental Protection Agency has developed rapid bioassessment protocols for sampling stream macroinvertebrates and analysis of macroinvertebrate data (Plafkin et al. 1989, Barbour et al. 1999) that should be useful in ERMN. These protocols advocates one of two stratified sampling approaches based instream habitat (Barbour et al. 1999). The “single habitat” protocol calls for sampling in riffle areas only because aquatic invertebrate assemblages tend to be most productive and diverse in these habitats, and because traditional sampling devices (e.g., D-frame kick nets, Hess samplers, etc.) are most effective in shallow, flowing water. However, riffle habitats are often not prevalent or easily discernable in the steep, cascading streams common in ERMN. In contrast, the “multiple habitat” approach involves sampling several different habitat types including cobble-bottoom areas, snags, macrophytes, and vegetated banks (Barbour et al. 1999). However, this approach requires estimates of the relative areas occupied by each habitat which either requires extensive measurements or a qualitative ranking that introduces a source of investigator bias. A more randomized approach such as that used by Snyder et al. (2002) in Delaware Water Gap National Recreation Area streams may be a modification worth considering because it provides repeatable and comparable data irrespective of stream type.

The other major question regarding sampling methods is whether to use field-based identification or laboratory based identification of specimens. The field based ID methods require considerably less time and expense, and sufficient taxonomic resolution of specimens is possible with field based methods as long a biologist with sufficient experience with the regional fauna is part of the field team. Lab methods provide greater flexibility because samples can be identified to various levels of taxonomic resolution and samples remain available for further analysis. However, increased handling and sample processing time associated with laboratory identifications increases the expense of the program dramatically.

***Limitations of Data and Monitoring*:** Timing of aquatic macroinvertebrate sampling is critical for obtaining comparable data, especially if sampling is conducted only once per year. Component macroinvertebrate species have complex and highly variable life cycles often including terrestrial stages. Consequently, macroinvertebrate assemblage metrics exhibit considerable natural variation across seasons. As a result, sampling windows tend to be relatively small and there is little flexibility in sampling schedules which can make scheduling field crews and other logistical matters difficult to accommodate. In addition, considerable investigator experience is required to identify organisms to levels beyond the ordinal stage in the field. As a result, protocols that require field identifications usually require a biologist with considerable taxonomic experience with the regional fauna. Moreover, for generic or species level identifications, samples usually need to be returned to the laboratory for processing which is time consuming and expensive. Finally, macroinvertebrate distributions within stream reaches are highly contagious and thus a large number of samples or complex stratification schemes are necessary to effectively characterize the assemblage.

***Key References*:**

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

DeShon, J.E. 1995. Development and application of the invertebrate community index (ICI). Pages 217-243 *in* Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making (W.S. Davis and T.P. Simon, editors). Lewis Publishers, Boca Raton, Florida.

Kerans, B. L. and J. R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecological Applications 4:768-785.

Lenat, D. R., L. A. Smock, and D. L. Penrose. 1980. Use of benthic macroinvertebrates as indicators of environmental quality. Pages 97-112 *in* Biological Monitoring for Environmental Effects (D. L. Worf, editor). D. C. Heath, Lexington, MA.

Resh, V. H. and J. K. Jackson. Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. Pages 195-223 in Freshwater Biomonitoring and Benthic Macroinvertebrates (D. M. Rosenberg and V. H. Resh, editors). Chapman and Hall, New York.

Rosenberg, D. M. and V. H. Resh. 1993. Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall Press, New York. 487pp.

Snyder, C. D., J. A. Young, D. R. Smith, and D. M. Lemarie. 2002. Influence of eastern hemlock (*Tsuga Canadensis*) forests on aquatic invertebrate assemblages in headwater streams. Canadian Journal of Fisheries and Aquatic Sciences 59:262-275.

Webster, J. R. 1983. The role of benthic macroinvertebrates in detritus dynamics of streams: a computer simulation. Ecological Monographs 49:51-72.

.

***Related Environmental Issues and Linked Vital Signs*:** Aquatic macroinvertebrates are sensitive to a wide range of instream, riparian, and landscape features that vary naturally and are themselves altered by human disturbance. In particular, stream channel characteristics (VS07), water quality (VS17 and VS17), stream hydrology (VS13), riparian vegetation (VS28), periphyton (VS42), landscape pattern (VS58) and land use change (VS57) are all linked to aquatic macroinvertebrate assemblage measures.

***Overall Assessment*:** Based on the proven ability of measures of aquatic macroinvertebrate assemblage structure and composition to discern impact and change, combined with the relatively high degree of power to assess change and the relatively low cost to sample, aquatic macroinvertebrates are probably the single best biological group to monitor to assess the health of small and mid-sized streams.

Level 1 ► Biological Integrity
Level 2 ► Focal Species or Communities
Level 3 ► Periphyton – Algae, diatoms, fungi, bacteria and protozoa (VS40)

***Brief Description*:** "Periphyton" refers to benthic algae, or algae that attach to substrate. For the most part, bioassessment programs incorporating periphyton use assemblage level metrics or indicator species to infer ecological condition. To a lesser extent, periphyton biomass (e.g., chlorophyll *a* concentrations) and productivity measures have been used.

***Significance/Justification*:** Benthic algae are the main primary producers in streams and therefore represent an important component of many stream food webs. In addition, periphyton assemblages often help stabilize substrata and provide habitat for many other organisms including bacteria, protozoans, and macroinvertebrates. Attributes that make periphyton good candidates for incorporation into the Vital Signs monitoring program include: 1) rapid reproduction and short life cycles making them valuable indicators of short-term impacts, 2) as primary producers, algae are most directly affected by physical and chemical changes in the environment, 3) algal assemblages have been shown to be sensitive to some stressors when other groups (e.g., macroinvertebrates and fish) were not, and 4) sampling for algae is relatively easy and inexpensive (Patrick 1973, Barbour et al. 1999). In addition, depending on the region, many algal species, especially diatoms, have been identified to have specific tolerances to various types of pollution strengthening the likelihood of establishing causal linkages between assemblage composition and specific stressors (Lowe 1974).

***Proposed Metrics*:** Many of the same indices (e.g., Shannon Diversity, Percent Community Similarity) and assemblage metrics (e.g., species richness) commonly used for other groups have also been used for periphyton assemblages to infer ecological condition. In addition, indices specific to periphyton have been developed such as the Pollution Tolerance Index (Lange-Bertalot 1979). Barbour et al. (1999) describes a suite of metrics that have been shown to be useful in inferring ecological condition and includes metrics such as species richness, percent sensitive diatoms, percent aberrant diatoms (diatoms with morphological anomalies), percent mobile diatoms, and the proportional representation of several key indicator species. In addition, estimates of periphyton biomass have also been used to in bioassessments, especially surveys designed to detect effects of nutrient enrichment or toxicity (Stevenson and Lowe 1986).

***Prospective Method(s) and Frequency of Measurement*:** There has been relatively little standardization in terms of periphyton sampling for bioassessment. However, Barbour et al. (1999) describe two rapid bioassessment protocols for periphyton that are a composite of techniques used in bioassessment programs in the States of Kentucky, Oklahoma, and Montana. The main difference between the two approaches is that one uses laboratory based identification and the other is a completely field-based protocol. Either approach would be useful although the completely field-based approach may be only sensitive to fairly large environmental changes.

***Limitations of Data and Monitoring*:** Although periphyton is found in virtually all streams, smaller headwater streams in forested landscapes are typically too shaded through much of the year to support many species, although those that do occur tend to be adapted to low light conditions. Thus, either bioassessments involving periphyton should be limited to mid-reach streams (stream orders 3-5) where significantly more light penetrates forest canopies and reaches stream bottoms, or, sampling needs to be confined to a very narrow sampling window in early spring just prior to canopy development when periphyton populations tend to reach maxima (Barbour et al. 1999). Moreover, as with aquatic macroinvertebrates, species composition of periphyton assemblages exhibit considerable seasonal variation, mainly in response seasonal changes in light and temperature patterns. As a result, either repeated sampling is required among seasons, or sampling must be conducted over a fairly narrow time windows to ensure comparability. In addition, periphyton tends to be patchily distributed within a stream reach, even within apparently uniform stream sections. Consequently, although sampling is relatively easy, a significant number of samples are required to ensure the assemblage is accurately represented, and taxonomic expertise is required in the field. It may be useful to consider an adaptive sampling approach (e.g., Smith et al. 2003) for this group. Finally, periphyton is extremely sensitive to scouring and consequently samples need to be collected during prolonged periods of stable flow. Frequently the best periods to sample periphyton in terms of community composition (i.e., early spring) are also the least predictable in terms of stream flow.

***Key References*:**
Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

DeShon, J.E. 1995. Development and application of the invertebrate community index (ICI). Pages 217-243 *in* Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making (W.S. Davis and T.P. Simon, editors). Lewis Publishers, Boca Raton, Florida.

Lange-Bertalot, H. 1979. Pollution tolerance as a criterion for water quality estimation. Nova Haedwigia 64:285-304.

Lowe, R.L. 1974. Environmental requirements and pollution tolerance of freshwater diatoms. U.S. Environmental Protection Agency, Environmental Monitoring Series, Cincinnati, Ohio.

Smith, D.R., R.F. Villella, and D.P. Lemarie. Application of adaptive cluster sampling to low-density populations of freshwater mussels. Environmental Ecological Statistics 10:7-15.

Stenvenson, R.J. and R.L. Lowe. 1986. Sampling and interpretation of algal patterns for water quality assessments. Pages 118-149 *in* Rationale for Sampling and Interpretation of Ecological Data in the Assessment of Freshwater Ecosystems (B.G. Isom, editor).

***Related Environmental Issues and Linked Vital Signs*:** Periphyton assemblages are sensitive to a wide range of instream, riparian, and landscape features that vary naturally and are themselves altered by human disturbance. In particular, stream channel characteristics (VS07), water quality (VS17 and VS17), stream hydrology (VS13), riparian vegetation (VS28), landscape pattern (VS58) and land use change (VS57) are all linked to periphyton measures. In turn, periphyton assemblage composition and productivity can potentially affect the composition and productivity of higher trophic groups including macroinvertebrates (VS41) and fish (VS44).

***Overall Assessment*:** Assessments using periphyton could add significantly to the vital signs monitoring program and they potentially offer diagnostic elements not obtained by aquatic macroinvertebrates or fish. However, there would also be considerable overlap in terms of information obtained with aquatic macroinvertebrates. Consequently periphyton might be considered a secondary group to monitor if sufficient resources are available.

Level 1 ► Biological Integrity
Level 2 ► Focal Species or Communities
Level 3 ► Fish Assemblages (VS44)

***Brief Description*:** "Fish assemblages" refers to measures of the structure and composition of the fish community.

***Significance/Justification*:** Fish are important components of most healthy stream ecosystems, occupying the top of the food web. Moreover, unlike other groups, the condition of fish populations is frequently of interest to the broad public due to their importance in terms of recreation and food. Even more importantly fish have numerous characteristics that are advantageous from a biological monitoring and assessment perspective which include: 1) they are relatively easy to collect and identify, 2) because they are among the longest lived species in streams and mobile, they are good indicators of long-term effects and broad habitat conditions, 3) the life histories and environmental requirements of most species are well known, and 4) they occupy positions throughout the aquatic food web and thus provide an integrateive view of watershed conditions (Karr 1986, Barbour et al. 1999). In addition, unlike aquatic macroinvertebrates and periphyton assemblages that exhibit wide natural variation seasonally, the relative abundance of fish species (excluding young-of-the-year individuals) remains relatively stable. As a result, fish sampling can occur over much broader sampling windows which allows more flexibility in terms of logistics.

Proposed Metrics: Two approaches to drawing inferences regarding ecological condition of streams are proposed. The first is a multivariate analysis approach that compares community composition within a stream reach from one time to the next based on the relative abundance of component taxa. This method uses traditional multivariate ordination or classification techniques such as Principal Components Analysis, Canonical Correlation Analysis, Discriminant Analysis, or K-means clustering (see Gauch 1982 for detailed synthesis of multivariate methods), and is useful because it allows evaluations of overall structure (groupings of species) as well as changes in individual species. The second approach is the multi-metric approach that has become increasingly common for bioassessments involving fish, especially in North America. This approach involves the calculation of numerous individual assemblage metrics, combining the information into a single score (i.e., index of biotic integrity or IBI), and comparison of the combined index score with the range of scores expected for streams of similar type in the region (Karr et al. 1986). Both integrated scores and values for individual metrics can be evaluated and monitored and have been shown to be useful assessment tools.

The metrics selected for inclusion into multi-metric approach depend on stream type and geographic region because expected scores for individual metrics vary in ecologically healthy streams. However, metrics usually include measures of species richness and composition, trophic composition, and fish abundance and condition (Karr 1987). Moreover, indices of biotic integrity (IBIs) have been developed for most regions of the country and many stream types (e.g., cold water streams versus warm water streams, highland streams versus coastal or piedmont streams, high-gradient versus low gradient, etc.) (see Miller et al. 1988 for review). Particularly relevant to ERMN are IBIs developed and tested for highland streams in Maryland

(Roth et al. 1998), the mid-Atlantic highlands (McCormick et al. 2001), and for small, cool water streams in the Appalachian Plateau of West Virginia (Leonard and Orth 1986).

***Prospective Method(s) and Frequency of Measurement*:** Because most species reproduce only once a year, populations tend to be stable throughout the year (if young-of-the-year fish are not considered). As a result, a single sampling during the relatively long period of base flow conditions is all that is required to adequately assess fish assemblages within a stream reach. Electrofishing has been shown to be the most effective sampling technique for collecting information on the broad fish community. Depending on stream size and depth profiles, a DC current backpack electroshocking unit, or a towable Pram electroshocking unit, operated by a team of either two or three individuals experienced with electoshocking techniques is needed to effectively sample small and mid-sized wadeable streams. The team should include a fish biologist with knowledge of the regional fauna so that species-level identifications can be made in the field. Alternatively, samples would need to be returned to the lab which not only is significantly more costly and time consuming but also can impact the resident fauna. Detailed methods for fish shocking protocols are described in Barbour et al. (1999).

***Limitations of Data and Monitoring*:** Because of natural barriers to movement like waterfalls and beaver dams, fish assemblages are naturally species poor in many smaller streams. For example, many of the streams draining steep terrains in Delaware Water Gap National Recreation Area support only a single species of fish (Ross et al. 2003) making assemblage level assessments meaningless. Thus, ecological assessments using fish should probably be limited to mid-reach streams. In addition, fish tend to be poor indicators of intermittent stresses because they will move from stream reaches during stressful times but return when conditions improve. Finally, relative to other groups, fish sampling requires more expensive and labor-intensive methods to effectively survey.

***Key References*:**
Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

Gauch, H.G., Jr. 1982. Multivariate Analysis in Community Ecology. Cambridge University Press, New York. 298 pp.

Karr, J.R. 1986. Assessing biological integrity in running waters: A method and its rationale. Illinois Natural History Survey Special Publication 5. Illinois Natural History Survey, Champaign, Illinois.

Leonard, P.M. and D.J. Orth. 1986. Application and testing of an index of biotic integrity in small, coolwater streams. Transactions of the American Fisheries Society 115:401-414.

McCormick, F.H., R.M. Hughes, P.R. Haufmann, A.T. Herlihy, D.V. Peck, and J.L. Stoddard. 2001. Development of an index of biotic integrity for the Mid-Atlantic Highlands region. Transactions of the American Fisheries Society 130:857-877.

Miller, D.L., P.M. Leonard, R.M. Hughes, J.R. Karr, P.B. Moyle, L.H. Schrader, B.A. Thompson, R.A. Daniel, K.D. Fausch, G.A. Fitzhugh, J.R. Gammon, D.B. Halliwell, P.L. Angermeier, and D.J. Orth. 1988. Regional applications of an Index of Biotic Integrity for use in water resource management. Fisheries 13:12-20.

Ross, R. M., R. M. Bennett, C. D. Snyder, J. A. Young, D. R. Smith, and D. P. Lemarie. 2003. Influence of eastern hemlock (Tsuga Canadensis L.) on fish community structure and function in headwater streams of the Delaware River basin. Ecology of Freshwater Fish 12:60-65.

Roth, N., M. Southerland, J. Chaillou, R. Klauda, P. Dazyak, S. Stranko, et al. 1998. Development of a fish index of biotic integrity. Environmental Monitoring and Assessment 51:89-106.

***Related Environmental Issues and Linked Vital Signs*:** Fish assemblages are influenced by a wide range of instream, riparian, and landscape features that vary naturally and are themselves altered by human disturbance. In particular, stream channel characteristics (VS07), water quality (VS17 and VS17), stream hydrology (VS13), riparian vegetation (VS28), periphyton (VS42), landscape pattern (VS58) and land use change (VS57) are all linked to fish assemblage measures.

***Overall Assessment*:** Fish assemblage measures can be an excellent indicator of ecological condition in streams if the resident fauna is well known and reference conditions have been documented. Fish sampling is more expensive than for other groups such as periphyton and macroinvertebrates, but it is easier to effectively characterize fish assemblages. Moreover, because they are longer lived, fish assemblage metrics provide a better indicator of long-term trends in ecological condition than other groups.

Level 1 ► Biological Integrity
Level 2 ► Focal Species or Communities
Level 3 ► Vernal Pond Amphibians (VS46)

***Brief Description*:** "Vernal Pond Amphibians" (VPA) refers to frogs (anurans) and salamanders (caudates) that breed in vernal ponds. Although in the truest sense, the term "vernal pond" refers exclusively to wetlands that fill with water in spring, the term is often more generally applied to all seasonal or semi-permanent wetlands that do not contain fish (Colburn 2004), and we adopt that broader definition here. In fact, in the eastern United States, many, if not most, temporary or seasonally-flooded wetlands actually fill in autumn soon after leaf off when deciduous trees cease transpiration, although they usually reach their maximum size and volume in spring.

***Significance/Justification*:** Concern over the status of amphibians has heightened in recent years due to increasing evidence of global and regional population declines, range reductions and extinctions (Wyman 1990). Although degradation of local habitat is implicated in many of the noted declines, they are not limited to highly degraded areas. Significant losses have been reported even in relatively pristine areas such as National Parks (Blaustein and Wake 1990), indicating the potential role of regional (e.g., acid deposition) and global (e.g., ozone depletion and climate change) stressors. In addition, because they require relatively undegraded aquatic and terrestrial habitats to complete their life cycles, as well as intact migration corridors between the two habitats, vernal pond amphibians are widely viewed as indicators of the condition of the larger forested ecosystem.

Proposed Metrics: Assemblage level measures such as species richness would be difficult and expensive to incorporate into long-term monitoring because accurate and precise measures would require repeated visits to each selected pond over a relatively long sampling interval, and require the use of a variety of sampling methods (visual encounter surveys, call counts, cover boards, etc.) that require significant expertise or training and are difficult to standardize when using multiple field crews (Heyer et al. 1994).

We propose using the number of wood frog (*Rana sylvatica*) and spotted salamander (*Ambystoma maculatum*) egg masses within selected ponds as vital sign metrics. Both species are ubiquitous, their eggs are relatively easy to identify and count, egg mass abundances of both species have been shown to strongly correlate with the number of breeding adults (_ and Patton), and they both occur in vernal ponds at approximately the same time and so sampling could be conducted concurrently. At the same time, the two species differ in many important life history traits such as mobility, longevity, and mating behaviors that make them vulnerable to different stressors.

***Prospective Method(s) and Frequency of Measurement*:** Complete censuses of the number of egg masses of both species laid in individual vernal pools are often possible and relatively rapid for smaller pools that contain low or moderate egg mass densities. However, sampling is advisable for larger wetlands and for wetlands with very high egg mass densities. Transect sampling (e.g., Snyder et al. 2005) should be used in larger ponds and high density ponds.

***Limitations of Data and Monitoring*:** Egg mass abundance data for both species are highly sensitive to annual variation in weather during and for some period proceeding breeding seasons (Semlitsch 2000), and consequently relatively long periods are required to assess trends. In addition, the timing of sampling will vary annually depending on when ponds thaw making scheduling the timing of field sampling more difficult. Finally, egg mass abundance of both species is highly dependant on hydroperiod. Thus, to the extent possible, hydroperiod should be considered a priori in the site selection process.

***Key References*:**
Blaustein, A. R. and D. B. Wake. 1990. Declining amphibian populations: A global phenomenon? Trends in Ecology and Evolution 5:203-204.

Colburn, E. A. 2004. Vernal Pools: Natural History and Conservation. McDonald and Woodward Publishing Company, Blacksburg, Virginia. 426pp.

Heyer, W. R., M. A. Donnelly, R. W. McDiarmid, L. C. Hayek, and M. S. Foster. Measuring and Monitoring Biological Diversity: Standard Methods for Amphibians. Smithsonian Institution Press, Washington, D.C. 364pp.

Semlitsch, R.D. 2000. Principles for management of aquatic-breeding amphibians. Journal of Wildlife Management 64:615-631.

Snyder, C. D., J. J. Julian, and J. A. Young. 2005. Assessement of ambystomatid salamander populations and their breeding habitats in Delaware Water Gap National Recreation Area. Report to the National Park Service. Final Report submitted to the National Park Service, Delaware Water Gap National Recreation Area.

Wyman, R. L. 1990. What's happening to the amphibians? Conservation Biology 4:350-352.

***Related Environmental Issues and Linked Vital Signs*:** Wood frog and spotted salamander egg mass abundances are affected by several key environmental variables that vary naturally at different scales. In particular, temperature and rainfall patterns (VS4), wetland hydrology (VS14), groundwater hydrology (VS15), wetland plants (VS23), and landscape pattern (VS58) are all important determinants of breeding pond selection and use by these two species. Some of these variables such as wetland hydrology and landscape pattern (e.g., relative pond isolation) can be accounted for *a priori* in the sampling design and site selection process. The other variables will need to be accounted for *post hoc* in the trend analysis phase.

***Overall Assessment*:** Measures of pond breeding amphibians can be sensitive indicators of overall ecosystem health. The sensitivity of this group of animals to local threats such as direct habitat destruction and disruption of migration corridors through roads and other changes in land use, combined with noted sensitivity to regional and global stressors make vernal pond amphibians a high priority group to monitor.

Level 1 ► Biological Integrity
Level 2 ► Focal Species or Communities
Level 3 ► Streamside Salamanders (VS47)

***Brief Description*:** Streamside salamanders have been identified as a strong candidate as a vital sign for tributary watersheds in the ERMN. An index using the streamside salamander community has been developed for the Mid-Atlantic Highlands. The index responds to multiple stressors and can be implemented by personnel with minimal training.

Significance/Justification: Most amphibian species require both terrestrial and aquatic habitats at various times of their life cycles, although some species spend considerably more time in truly aquatic habitats. Streamside salamanders occupy the functional role of aquatic vertebrates in the upper reaches of tributary watersheds, where fishes are often absent. Although the community of plethodontid salamander species is modest in size, the presence / absence of selected species correspond with patterns of landscape disturbance. An index has been developed and tested throughout the Mid-Atlantic Highlands; Streamside Plethodontid Assessment R (SPAR, Rocco et al. 2004), corresponding to the ERMN.

***Proposed Metrics*:** The SPAR index has been calibrated for different regions of the Mid-Appalachian Highlands, and thus, is readily available for use. Also, initial results of bioaccumulation studies, have shown that concentrations of contaminants can be measured in streamside salamanders by qualified laboratories (G. Rocco, pers. comm.), adding another potential use of this taxon as a vital sign in the ERMN. Density and abundance measures derived from captures may also serve as vital signs, although additional calibration is necessary.

***Prospective Method(s) and Frequency of Measurement*:** Standard sampling protocols have been developed (SPAR, Rocco et al. 2004). These protocols have been tested and found to be appropriate for use with trained volunteers. The sampling period is fairly broad (spring through autumn). Relatively few plots (usually 3-5, 2 m x 2 m) are sampled per 1 km reach of stream to generate data for the SPAR index. An automated computation process has been developed to simplify data analysis (G. Rocco, pers. comm.). Sampling can be conducted in conjunction with aquatic macroinvertebrate sampling.

***Limitations of Data and Monitoring*:** Some training is required to conduct SPAR sampling, but this does not limit the utility of the technique.

***Key References*:**

Rocco, G. L., R. P. Brooks, and J. T. Hite. 2004. Stream plethodontid assemblage response (SPAR) index: development, application, and verification in the MAHA. Final Report. U.S. Environmental Protection Agency, STAR Grants Program, Washington, DC. Rep. No. 2004-01. Penn State Cooperative Wetlands Center, University Park, PA. 33pp+figs& app.

***Related Environmental Issues and Linked Vital Signs*:** Streamside salamanders appear to be responsive to multiple stressors in tributary watersheds. These amphibian taxa provide another dimension to understanding the response of biological communities to stressors in these systems, much like aquatic macroinvertebrates and fish.

***Overall Assessment*:** An established index using streamside salamanders has been developed and tested. The SPAR index is suitable for use as a vital sign.

Level 1 ► Ecosystem Pattern and Process
Level 2 ► Land Cover/Land Use
Level 3 ► Landscape Change and Pattern (VS57, 58, tributary)

***Brief Description*:** "Landscape Pattern" refers to the states and distribution of the various dominant cover types, as they exist within a landscape mosaic. In addition to the current pattern, historic patterns (Braun 1950) should be considered, as well as trends and changes in landscape patterns. These changes can be useful **indicators** of the natural and human-caused forces acting upon the landscape (Alig and Butler 2004). Turner et al. (2003) examined the landscape-level changes in the Appalachian region (including much of the ERMN) and found that during the four-decade interval from 1950 to 1990, the amount of forest cover increased and fragmentation decreased, but they cautioned that recent housing development in the region may offset many of these gains. Human impacts are a critical element in the changing landscapes of the ERMN and Ritters et al. (2000) indicate that land cover information provides a mechanism to place humans into ecological assessments.

Significance/Justification: Human activities are primary drivers in landscape-level changes in the Appalachians (Ritters et. al. 2000). Given that 70 – 80 percent of the landscape falls into the tributary watershed category, most activities affect this watershed component (Brooks et al. this document). Historical occurrences such as agricultural clearing, agricultural abandonment, timbering, surface mining, forest fire control, predator eradication, hunting regulation, insect and disease introductions and urban sprawl are all examples of how humans have contributed to landscape-level changes over the last hundred and fifty years. ERMN parks are in-effect islands within an ever-changing mosaic of land, and changes outside the ERMN parks can potentially affect the ecological properties within the parks (Brosofske et. al. 1999). Changes are particularly relevant to tributary watersheds where human activities outside of NPS land can affect downstream ecosystems located within NPS land. Roads have a particularly significant fragmenting effect on terrestrial ecosystems (Forman 2000; Trombulak and Fressell 2004). Changes in landscape pattern can alter habitat for neotropical birds, mammals (Dijak and Thompson 2000) and forest wetlands (Gibbs 2000). Because land use patterns surrounding the ERMN are changing and these changes have the potential for altering the ecological characteristics within the parks, it is important that park managers be aware of this process and how it is likely to affect them.

***Proposed Metrics*:** Landscape ecology is a field that uses spatial analysis methods to evaluate the pattern of various land cover types at different spatial scales. Metrics include the proportion of a given landscape occurring in a particular cover type and indices of patchiness, fragmentation, connectivity etc. In addition, changes in linear features such as streams and riparian zones, while not occupying large surface areas, have a profound effect on aquatic resources. These metrics can be used to compare among landscapes or to observe temporal changes in a single landscape.

***Prospective Method(s) and Frequency of Measurement*:** Spatial analysis methods begin with imagery (aerial photography, satellite images, etc.) and databases (USGS topographic information, ownership, etc.). Image information requires interpretation in order to determine what the visual information represents. Interpretation can be facilitated by image enhancing methods, such as digital color transformations. The resulting information is used to create a geographic information system (GIS) that incorporates multiple layers of spatial information, such as land use, ownership, cover type, topography, etc. Software packages are available that provide powerful tools for organizing, interpreting and displaying the information. Spatial statistics can be used to analyze the data (Gardner et. al. 1987), and models constructed using information from known landscapes can be used to predict the states of other landscapes (Trombulak and Frissell 2004).

Limitations of Data and Monitoring**:** A substantial amount of landscape-level information currently exists, much of which is public record, and therefore **inexpensive** to acquire. The problem with many available sources of images or spatial data is that they must be adapted to the specific use required (e.g. ERMN parks). The detail, scale and type of imagery may not suit the specific purpose of the ERMN, requiring that new and expensive data need to be gathered. The development of a system-wide GIS can be a daunting task, requiring that either contractors or trained NPS employees complete the work. Furthermore, as Li and Wu (2004) warn, landscape analysis often falls short of meeting its high expectations due to conceptual flaws in pattern analysis, inherent limitations of landscape indices and improper use of pattern indices.

Key References**:**
Alig, R.J. & Butler, B.J. 2004. Area changes for forest cover types in the United States, 1952 to 1997, with projections to 2050. USDA Forest Service, Gen. Tech. Rep. PNW-GTR-613, p. 106.

Braun, E. L. 1950. Deciduous Forests of Eastern North America. The Blakiston Co., Philadelphia, PA.

Brosofske, K.D., Chen, J., Crow, T.R., & Saunders, S.C. 1999. Vegetation responses to landscape structure at multiple scales across a northern Wisconsin, USA pine barrens landscape. Plant ecology, 143: 203-218.

Dijak, W.D. & Thompson, F.R. 2000. Landscape and edge effects on the distribution of mammalian predators in Missouri. Journal of Wildlife Management, 64(1): 209-216.

Forman, R.T.T. 2000. Estimate of the area affected ecologically by the road system in the United States. Conservation Biology, 14(1): 31-35.

Gardner, R.H., Milne, B.T., Turner, M.G., & O'Neill, R.V. 1987. Neutral models for the analysis of broad-scale landscape pattern. Landscape Ecology, 1(1): 19-28.

Gibbs, J.P. 2000. Wetland loss and biodiversity conservation. Conservation Biology, 14(1): 314-317.

Ritters, K.H., Wickham, J.D., Vogelman, J.E., & Jones, K.B. 2000. National land-cover data. Ecology, 81: 604

Trombulak, S.C. & Frissell, C.A. 2004. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology, 14(1): 18-30.

Turner, M.G., Pearson, S.M., Bolstad, P. & Wear, D.N. 2003. Effects of land-cover change on spatial pattern of forest communities in the Southern Appalachian Mountains (USA). Landscape Ecology, 18: 449-464.

***Related Environmental Issues and Linked Vital Signs*:** Landscape pattern is related to many environmental issues, particularly ones having to do with anthropogenic effects, such as pollution, land use, settlement, etc. Landscape patterns are linked with almost all the vital signs identified for ERMN parks. Atmospheric and climatic patterns (VS1- VS4) vary across the landscape, and these factors, in turn, create patterns in vegetation and land use. Geology and soils (VS6- VS12) also contribute to landscape patterns, as well as do hydrologic features (VS14, VS15). Because human activities often are involved in the introduction of invasive species, and human habitation is part of the landscape pattern, the pattern of introduction of invasive species (VS18) often the follows patterns of human activity (transportation, settlement, etc.). Plant and animal communities (VS 20- VS48) are specifically adapted to their environment, which changes across the landscape. Visitor usage (VS54) can locally alter an ecosystem, therefore imposing an anthropogenic pattern on the landscape. Finally, bio-productivity and nutrient dynamics (VS59, VS61) are specifically linked to the landscape pattern. In short, the pattern that exists on the landscape is a reflection of the sum of the abiotic, biotic and anthropogenic factors that interact over it.

***Overall Assessment*:** Landscape pattern is a result of the interaction of numerous factors (historic and present). ERMN parks are themselves part of a larger landscape, and are affected by actions that take place beyond their boundaries. The discipline of landscape ecology has been developing in recent years and involves using imagery, data, technology and statistical tools to analyze and interpret spatial information. ERMN managers can use these methods to assess current conditions in their parks as they relate to the larger landscape. Use of this tool may enable managers to anticipate changes and take remedial actions, when necessary.

Level 1 ► Ecosystem Pattern and Process
Level 2 ► At-risk Biota
Level 3 ► Park-specific Threatened, Endangered or Indicator Species (VS99)

***Brief Description*:** As public resource agencies, parks have an obligation to address conservation issues related to **state and federal threatened and endangered species, and species of special concern**, such as those that are vulnerable to various stressors (e.g., Noss 1990). In addition, there are selected species from a variety of taxa that may serve as broadly-defined **indicator species** (e.g., McKenzie et al. 1992). For example, carnivores like river otter or habitat-altering species such as beaver may serve as keystone species that influence the ecological integrity of biological communities.

Significance/Justification: Species such as mink, river otter and northern pike are carnivores at the top of food webs, thus, they are sensitive to accumulation of contaminants (e.g., heavy metals, PCBs) that enter aquatic ecosystems. If such contaminants are suspected or possible, and sources of tissue samples are readily available (e.g., fur trapping, road kills, recreational fishing), then a modest monitoring program may justified. Also, river otter can be an attractant for visitors and recreationists, so for some units, periodic presence / absence and/or density surveys may be warranted. In selected units, species of conservation concern, such as water shrews, rare fishes, dragonflies, spotted and bog turtles, or uncommon orchids may warrant the use of targeted protocols to confirm their existence. The presence of beaver can significantly alter aquatic and vegetation features of tributary watersheds, with resultant habitat and/or economic damage. Thus, monitoring their presence and the extent of areas affected may be warranted.

***Proposed Metrics*:** For bioaccumulation studies, the concentration of suspected contaminants in target tissues (e.g., fat, reproductive organs) can be measured by qualified laboratories. Density measures derived from sampling protocols, observed sign or captures of individuals may suffice for the other purposes.

***Prospective Method(s) and Frequency of Measurement*:** Standard methods are available for most of the suggested approaches to monitoring aquatic flora and fauna, especially for state and federal threatened and endangered species (e.g., Barbour et al. 1999). Recovery plans for such species, if available, should be consulted to obtain appropriate sampling techniques and to learn about regional goals for recovery. The expected frequency of measurement is likely to be seasonal or annual for most species. Inexpensive field monitoring protocols have been developed for river otters and beaver for the Delaware Water Gap unit (Swimley et al. 1998, Serfass and Brooks 1998).

***Limitations of Data and Monitoring*:** The primary limitation to using rare species and indicator species as vital signs is the low density of many species which can translate to limited data availability. Some indicator species, however, can be common and are readily observable.

***Key References*:**

Barbour, M.T., J. Gerritsen, B.D. Synder, and J.B. Stribling. 1999. Rapid Bioassessment Protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates, and fish, Second Edition. U.S. Environmental Protection Agency, Office of Water, EPA/841-B-99-002, Washington, DC.

McKenzie, D. H., D. E. Hyatt, and V. J. McDonald (eds.). 1992. Ecological indicators. Proc. Int. Symp. Ecological Indicators, 16-19 October 1990, Fort Lauderdale, FL. Elsevier Science Publishers, Ltd., Esses, England. Vols. 1 & 2.

Noss, R. F. 1990. Indicators for monitoring biodiversity: a hierarchical approach. Conserv. Biol. 4(4):355-364.

Serfass, T. S., and R. P. Brooks. 1998. Wetlands mammals. Pages 350-358 in S. K. Majumdar, E. W. Miller, and F. J. Brenner (eds.). Ecology of wetlands and associated systems. Penn. Acad. Sci., Easton, PA. 685pp.

Swimley, T. J., T. L. Serfass, R. P. Brooks, and W. M. Tzilkowski. 1998. Predicting river otter latrine sites in Pennsylvania. Wildlife Society Bulletin 26(4):836-845. 1998.

***Related Environmental Issues and Linked Vital Signs*:** Rare aquatic species, although few in number, can be important as sentinels (mink, river otter) and agents (beaver) of environmental change. Rare flora may be sought after by collectors, so collection permits and take issues may arise and need to be monitored.

***Overall Assessment*:** Use of rare species and indicators species as vital signs is recommended in selected situations. Their use may be appropriate where park units serve as refugia for aquatic species that are legally harvested inside or outside parks, and where bioaccumulating contaminants are suspected to be present. Specific park units may need to collaborate regionally with other public and private conservation organizations to ensure the survival of species of concern.

LITERATURE CITED

Ainslie, W. B., R. D. Smith, B. A. Pruitt, T. H. Roberts, E. J. Sparks, L. West, G. L. Godshalk, and M. V. Miller. 1999. A Regional Guidebook for Assessing the Functions of Low Gradient, Riverine Wetlands in Western Kentucky. WRP-DE-17, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Albanese, B., P. L. Angermeier, and S. Dorai-Raj. 2004. Ecological correlates of fish movement in a network of Virginia streams. Canadian Journal of Fisheries and Aquatic Sciences 61:857-869.

Allan, J. D. 1995. Stream Ecology: Structure and Function of Running Waters. Chapman and Hall Publishers, London England. 388pp.

Anderson, N. H. 1966. Depressant effect of moonlight on activity of aquatic insects. Nature 209:319-320.

Anderson, Stanley H., and Kevin. J. Gutzwiller. 1994. Habitat evaluation methods. Pages 592-606 in Research and management techniques for wildlife and habitats (Theodore A. Bookhout, Ed.). The Wildlife Society, Bethesda, MD.

Arcement, G.J., Jr. and V.R. Schneider. 1989. Guide for selecting Manning's roughness coefficients for natural channels and flood plains. U.S. Geological Survey Water-Supply Paper 2339, U.S. Government Printing Office, Washington, DC.

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols or use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, 2nd ed. EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Beltman, D. J., W. H. Clements, J. Lipton, and D. Cacela. 1999. Benthic invertebrate metals exposure, accumulation, and community-level effects downstream from a hard-rock mine site. Environmental Toxicology and Chemistry 18:299-307.

Benke, A. C., I. Chaubey, G. M. Ward, and E. L. Dunn. 2000. Flood pulse dynamics of an unregulated river floodplain in the southeastern U.S. coastal plain. Ecology 81:2730-2741.

Bilby, R. E., J. T. Heffner, B. R. Frasen, and J. W. Ward. 1999. Effects of immersion in water on deterioration of wood from five species of trees used for habitat enhancement projects. Journal of North American Fisheries Management 19:687-695.

Bilby, R. E., and G. E. Likens. 1979. Effect of hydrologic fluctuations on the transport of fine particulate organic carbon in a small stream. Limnology and Oceanography 24:69-75.

Boling, R. H. Jr., E. E. Goodman, J. A. Van Sickle, and O. J. Zimmer. 1975. Toward a model of detritus processing in a woodland stream. Ecology 56:141-151.

Boltz, J. M., and J. R. Stauffer. 1989. Fish assemblages of Pennsylvania wetlands. Pages 158-170 in J. M. Majmdar, R. P. Brooks, F. J. Brenner, and R. W. Tiner. Wetlands ecology and conservation: emphasis in Pennsylvania. Pennsylvania Academy of Science, Easton, PA. 395pp.

Bosco, J., and J. A. Perry. 2000. Drift and benthic invertebrate responses to stepwise and abrupt increases in non-scouring flow. Hydrobiologia 436:191-208.

Boto, K. G., and J. W. H. Patrick. 1978. Role of wetlands in removal of suspended sediments. Pages 479-489 in P. E. Greeson, J. R. Clark, and J. E. Clark, editors. Wetland functions and values: The state of our understanding. Proc. Nat'l Symp. on Wetlands. American Water Resources Association, Minneapolis, MN.

Boward, D. M., P. F. Kazyak, S. A. Stranko, M. K. Hurd, and P. P. Prochaska. 1999. From the mountains to the sea: the state of Marylands' freshwater streams. EPA 903-R-99-023. Maryland Dep. Natural Resources, Monitoring and Nontidal Assessment Division, Annapolis, MD. 54pp.

Brenner, J. J. 2000. Use of constructed wetlands for acid mine drainage abatement and restoration. Water Science and Technology 44:11-12.

Briers, R. A., H. M. Cariss, and J. H. R. Gee. 2002. Dispersal of adult stoneflies (Plecoptera) from upland streams draining catchments with contrasting land use. Archiv fur Hydrobiologie 155:627-644.

Brinson, M.M., A.E. Lugo and S. Brown. 1981. Primary productivity, consumer activity, and decomposition in freshwater wetlands. Annual Review of Ecology and Systematics 12:123-161

Brinson, M. M. 1990. Riverine Forests. Pages 87-141 in A. E. Lugo, M. M. Brinson, and S. Brown, editors. Forested Wetlands: Ecosystems of the World. Elsevier, Amsterdam.

Brinson, M. M. 1993. A hydrogeomorphic classification for wetlands. Technical Report WRP-DE-4, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Brinson, M. M., F. R. Hauer, L. C. Lee, W. L. Nutter, R. D. Rheinhardt, R. D. Smith, and D. Whigham. 1995. A guidebook for application of hydrogeomorphic assessments to riverine wetlands (Operational draft). Wetlands Research Program Technical Report WRP-DE-11, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Brinson, M. M. and A. I. Malvarez. 2002. Temperate freshwater wetlands: types, status, and threats. Environmental Conservation 29:15-133.

Brooks, R. P., D. A. Devlin, and J. Hassinger (with M. C. Brittingham and G. A. Hoover). 1993. Wetlands and Wildlife. College of Agricultural Sciences, Pennsylvania State University, PA. 55pp.

Brooks, R. P., and D. J. Prosser. 1995. Wildlife Habitat Suitability Models. CWC Report 95-1, Penn State Cooperative Wetlands Center, University Park.

Brooks, R. P., T. J. O'Connell, D. H. Wardrop, and L. E. Jackson. 1998. Towards a regional index of biological integrity: the examples of forested riparian ecosystems. Environ. Monit. Assmt. 51:131-143.

Brooks, R. P. (ed.). 2004. Monitoring and Assessing Pennsylvania Wetlands. Final Report for Cooperative Agreement No. X-827157-01, between Penn State Cooperative Wetlands Center, Pennsylvania State University, University Park, PA and U.S. Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Washington, DC. Rep. No. 2004-3 Penn State Cooperative Wetlands Center, University Park, PA.

Brooks, Robert P., Denice Heller Wardrop, and Joseph A. Bishop. 2004. Assessing wetland condition on a watershed basis in the Mid-Atlantic region using synoptic land cover maps. Environmental Monitoring and Assessment 94:9-22.

Brookshire, E. N. J. and K. A. Dwire. 2003. Controls on patterns of coarse organic particle retention in headwater streams. Journal of the North American Benthological Society 22:17-34.
Brown, R. G. 1988. Effects of wetland channelization on runoff and loading. Wetlands 8:123-133.

Brown, S. L. 1990. Structure and dynamics of basin forested wetlands in North America. Pages 171-199 in A. E. Lugo, M. M. Brinson, and S. Brown, editors. Forested Wetlands: Ecosystems of the World. Elsevier, Amsterdam.

Bryce, S. A., D. P. Larsen, R. M. Hughes, P. R. Kaufmann. 1999. Assessing relative risks to aquatic ecosystems: a Mid-Appalachian case study. J. American Water Resources Assoc. 35:23-26.

Carline, R. F., W. E. Sharpe, and C. J. Gagen. 1992. Changes in fish communities and trout management response to acidification of streams in Pennsylvania. Fisheries 17:33-38.

Carter, V. 1986. An Overview of the Hydrologic Concerns Related to Wetlands in the United States. Canadian Journal of Botany 64:364-374.

Castelle, A. J., A. W. Johnson, and C. Conolly. 1994. Wetland and stream buffer size requirements-A review. Journal of Environmental Quality 23:878-882.

Church M. 2002. Geomorphic thresholds in riverine landscapes. Freshwater Biology 47:541-557

Cirmo, C. P., C. T. Driscoll, and K. Bowes. 2000. Chemical fluxes from sediments in two adironkack wetlands: Effects of an acid-neutralization experiment. Soil Science Society of America Journal 64:790-799.

Cole, C. A., R. P. Brooks, and D. H. Wardrop. 1997. Wetland hydrology as a function of hydrogeomorphic (HGM) subclass. Wetlands 17:456-467.

Cole, C. A., and R. P. Brooks. 2000. A comparison of the hydrologic characteristics of natural and created mainstem floodplain wetlands in Pennsylvania. Ecological Engineering 14(3):221-231.

Collier, K. J. and B. J. Smith. 1998. Dispersal of adult caddisflies (Trichoptera) into forests alongside three New Zealand streams. Hydrobiologia 361:53-65.

Cooper, E. L. 1983. Fishes of Pennsylvania and the northeastern United States. Pennsylvania State University Press, University Park, PA. 243pp.

Cooper, J. R., J. W. Gilliam, and T. C. Jacobs. 1986. Riparian areas as a control of nonpoint pollutants. Pages 166-190 in D. L. Correll, editor. Watershed Research Perspectives. Smithsonian Institute Press, Washington DC.

Cooper, J. R., and J. W. Gilliam. 1987. Phosphorus redistribution from cultivated fields into riparian areas. Soil Science Society of America Journal 51:1600-1604.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. FWS/OBS-79/31, U.S. Fish and Wildlife Service, Washington D.C.

Cronan, C. S., J. T. Piampiano, and H. H. Patterson. 1999. Influence of land use and hydrology on exports of carbon and nitrogen in a Maine river basin. Journal of Environmental Quality 28:953-961.

Croonquist, M. J., and R. P. Brooks. 1991. Use of avian and mammalian guilds as indicators of cumulative impacts in riparian-wetland areas. Environ. Manage. 15(5):701-714.

Croonquist, M. J., and R. P. Brooks. 1993. Effects of habitat disturbance on bird communities in riparian corridors. J. Soil Water Conserv. 48(1):65-70.

Crumpton, W. G., T. M. Isenhart, and S. W. Fisher. 1993. Fate of non-point source nitrate loads in freshwater wetlands: Results from experimental wetland mesocosms. Pages 283-291 in G. A. Moshiri, editor. Constructed Wetlands for Water Quality Improvement. Lewis Publishers, Ann Arbor, Michigan.

Cuffney, T. F. 1988. Input, movement and exchange of organic matter within a subtropical coastal blackwater river-floodplain system. Freshwater Biology 19:305-320.

Cuffney, T. F, and J. B. Wallace. 1989. Discharge-export relationships in headwater streams: the influence of invertebrate manipulations and drought. Journal of the North American Benthological Society 8:331-341.

Cummins, K. W., R. C. Petersen, F. O. Howard, J. C. Wuycheck, and V. I. Holt. 1973. The utilization of leaf letter by stream detritivores. Ecology 54:336-345.

Cummins, K. W. and M. J. Klug. 1979. Feeding ecology of stream invertebrates. Annual Review of Ecology and Systematics 10:147-172.

Davidsson, T. E., and M. Stahl. 2000. The influence of organic carbon on nitrogen transformations in five wetland soils. Soil Science Society of America Journal 64:1129-1136.
Demissie, M., and A. Khan. 1993. Influence of wetlands on streamflow in Illinois. Contract Report 561, Illinois State Water Survey, Champaign, Illinois.

DiNicola, D. M. and M. G. Stapleton. 2002. Impact of acid mine drainage on benthic communities in streams: the relative roles of substratum vs. aqueous effects. Environmental Pollution 119:303-315.

Dosskey, M. G., and P. M. Bertsch. 1994. Forest sources and pathways of organic matter transport to a blackwater stream: A hydrologic approach. Biogeochemistry 24:1-19.

Dudgeon, D. 1990. Seasonal dynamics of invertebrate drift in a Hong Kong stream. Journal of Zoology 222:187-196.

Euliss Jr., N. H., and D. M. Mushet. 1996. Water-level fluctuation in wetlands as a function of landscape condition in the prairie pothole region. Wetlands 16:587-593.

Faust, S. D. 1983. Total alkalinity versus buffer value (capacity) as a sensitivity indicator for fresh waters receiving acid rain. Journal of Environmental Science and Health 18:701-711.

Forman, R. T. 1995. Land mosaics: the ecology of landscapes and regions. Cambridge University Press. 632pp.

Forester, D. C. and D. V. Lykens. 1991. Age structure in a population of red-spotted newts from the Allegheny Plateau of Maryland. Journal of Herpetology 25:373-376.

Freeland, J. A., J. L. Richardson, and L. A. Foss. 1999. Soil indicators of agricultural impacts on northern prairie wetlands: Cottonwood Lake Research Area, North Dakota, USA. Wetlands 19:56-64.

Frissell CA, Liss WJ, Warren CE, Hurley MD. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10:199-214.

Georgian, T. and J. B. Wallace. 1983. Seasonal production dynamics of a guild of periphyton-grazing insects in a southern Appalachian stream. Ecology 64:1236-1248.

Gerhardt, A. 1993. Review of impact of heavy metals on stream invertebrates with special emphasis on acid conditions. Water, Air, and Soil Pollution 66:289-314.

Gibbs, J. P. 1993. Importance of small wetlands for the persistence of local populations of wetland-associated animals. Wetlands 13(1):25-31.

Giberson, D. J. and D. Caissie. 1998. Stream habitat hydraulics: Interannual variability in three reaches of Catamaran Brook, New Brunswick. Canadian Journal of Fisheries and Aquatic Sciences 55:485-494.

Gilliam, J. W. 1994. Riparian wetlands and water quality. Journal of Environmental Quality 23:896-900.

Gosselink, J. G., and R. E. Turner. 1978. The Role of Hydrology in Freshwater Wetland Ecosystems. Pages 63-78 in R. E. Good, D. F. Whigham, and R. L. Simpson, editors. Freshwater Wetlands Ecological Processes and Management Potential. Academic Press.

Gregory, S. V., F. J. Swanson, W. A. McKee, and K. W. Cummins. 1991. An ecosystem perspective of riparian zones. BioScience 41:540-551.

Griffity, J. S., Jr. 1974. Utilization of invertebrate drift by brook trout (Salvelinus fontinalis) and cutthroat trout (Salmo clarki) in small streams in Idaho. Transactions of the American Fisheries Society 103:440-447.

Groffman, P. M. 1994. Denitrification in freshwater wetlands. Current Topics in Wetland Biogeochemistry 1:15-35.

Guasch, H., N. Ivorra, V. Lehmann, M. Paulsson, M. Real, and S. Sabater. 1998. Community composition and sensitivity of periphyton to atrazine in waters: the role of environmental factors. Journal of Applied Phycology 10:203-213.

Gurnell, A. M. and C. Linstead. 1998. Interactions between large woody debris accumulations, hydrologic processes, and channel morphology in British headwater rivers. Hydrology in a Changing Environment 1:381-396.

Harmon, M. E., J. F. Franklin, F. J. Swanson, P. Sollins, S. V. Gregory, J. D. Lattin, N. H. Anderson, S. P. Cline, N. G. Aumen, J. R. Sedell, G. W. Lienkaemper, K. C. Jr., and K. W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15:133-302.

Hart, D. D. and C. T. Robinson. 1990. Resource limitation in a stream community: Phosphorus enrichment effects on periphyton and grazers. Ecology 71:1494-1502.

Hemond, H. F., and J. Benoit. 1988. Cumulative impacts on water quality functions of wetlands. Environmental Management 12:639-653.

Herlihy, A. T., D. P. Larsen, S. Gl Paulsen, N. S. Urquhart, and B. J. Rosenbaum. 2000. Designing a spatially balanced, randomized reach selection process for regional stream surveys: The EMAP Mid-Atlantic pilot study. Environmental Monitoring and Assessment 63:95-113.

Hobbie, S. E. 1992. Effects of plant species on nutrient cycling. TREE 7:336-339.

Hughes, N. F. A model of habitat selection by drift-feeding stream salmonids at different scales. Ecology 79:281-294.

Hupp, C. R., M. D. Woodside, and T. M. Yanosky. 1993. Sediment and trace element trapping in a forested wetland, Chickahominy River, Virginia. Wetlands 13:95-104.

Iversen, T. M., J. Thorup, and J. Skriver. 1982. Inputs and transformation of allochthonous particulate organic matter in a headwater stream. Holarctic Ecology 5:10-19.

Jackson, J. K. and S. G. Fisher. 1986. Secondary production, emergence, and export of aquatic insects of a Sonoran desert stream. Ecology 67:629-638.

Jacoby, J. M., M. R. Seeley, E. B. Welch, and R. R. Horner. 1990. Responses of periphyton to changes in current velocity, suspended sediment, and phosphorus concentration. Freshwater Biology 42:215-232.

Joensuu, S. 1997. Factors affecting sediment accumulation in sedimentation ponds. Pages 297-311 in C. C. Trettin, M. F. Jurgensen, D. F. Grigal, M. R. Gale, and J. K. Jeglum, editors. Northern Forested Wetlands: Ecology and Management. Lewis Publishers, New York, NY.

Johnston, C. A. 1991. Sediment and nutrient retention by freshwater wetlands: Effects on surface water quality. Critical Reviews in Environmental Control 21:491-565.

Johnston, C. A. 1993a. Material fluxes across wetland ecotones in northern landscapes. Ecological Applications 3:424-440.

Johnston, C. A. 1993b. Mechanisms of wetland-water quality interaction. Pages 293-299 in G. A. Moshiri, editor. Constructed Wetlands for Water Quality Improvement. Lewis Publishers, MI.

Johnston, C. A., G. D. Bubenzer, G. B. Lee, F. W. Madison, and J. R. McHenry. 1984. Nutrient trapping by sediment deposition in a seasonally flooded lakeside wetland. Journal of Environmental Quality 13:283-290.

Jones, J. B. and P. J. Mulholland. 2000. Streams and ground waters. Academic Press, San Diego, California, 425pp.

Jones, J. B., Jr., and L. A. Smock. 1991. Transport and retention of particulate organic matter in two low-gradient headwater streams. Journal of the North American Benthol. Society 10:115-126.

Jordan, T. E., D. L. Correll, and D. E. Weller. 1993. Nutrient interception by a riparian forest receiving inputs from adjacent cropland. Journal of Environmental Quality 22:467-473.

Jordan, T. E., D. L. Correll, and D. E. Weller. 1997. Relating nutrient discharges from watersheds to land use and streamflow variability. Water Resources Research 33(11):2579-2590.

Junk, W. J., and R. L. Welcomme. 1990. Floodplains. Pages 491-524 in B. C. P. e. al., editor. Wetlands and Shallow Continental Water Bodies. SPB Academic Publishing, The Hague, The Netherlands.

Jurik, T. W., S.-C. Wang, and A. G. v. d. Valk. 1994. Effects of sediment load on seedling emergence from wetland seed banks. Wetlands 14:159-165.

Kadlec, R. H. 2001. Phosphorus dynamics in event driven wetlands. Pages 365-391 in J. Vymazal, editor. Transformations of Nutrients in Natural and Constructed Wetlands. Backhuys Publishers, Leiden, The Netherlands.

Karr, J. R. 1999. Defining and measuring river health. Freshwater Biology 41: 221-234.

Karr, J. R and E. W. Chu. 1999. Restoring Life in Running Waters: Better Biological Monitoring. Island Press, Washington, D.C. 206 pp.

King, R. S., M. E. Baker, D. F. Whigham, D. E. Weller, T. E. Jordan, P. F. Kazyak, and M. K. Hurd. 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15:137–153.

Kirchner, J. W. 1992. Heterogeneous geochemistry of catchment acidification. Geochimica et Cosmochimica Acta 56:2311-2327.

Kittle, D. L., J. B. McGraw, and K. Garbutt. 1995. Plant litter decomposition in wetlands receiving acid mine drainage. Journal of Environmental Quality 24:301-306.

Klemm, D. J., K. A. Blocksom, F. A. Fulk, A. T. Herlihy, R. M. Hughes, P. R. Kaufmann, D. V. Peck, J. L. Stoddard, W. T. Thoeny, and M. B. Griffith. 2003, Development and evaluation of a macroinvertebrate biotic integrity index (MBII) for regionally assessing Mid-Atlantic Highlands Streams. Environmental Management 31(5):656-669.

Lake PS. 2000. Disturbance, patchiness, and diversity in streams. Journal of the North American Benthological Society 19:573-592.

Lonzarich, D. G., M. R. Lonzarich, and M. L. Warren, Jr. 2000. Effects of riffle length on short-term movement in fishes among pools. Canadian Journal of Fisheries and Aquatic Sciences 57:1508-1514.

Majumdar, S. K., R. P. Brooks, F. J. Brenner, and R. W. Tiner. 1989. Wetlands ecology and conservation: emphasis in Pennsylvania. Pennsylvania Academy of Science, Easton, PA. 395pp.

Margolis, B. E., M. S. Castro, and R. L. Raesly. 2001. The impact of beaver impoundments on the water chemistry of two Appalachian streams. Canadian Journal of Fisheries and Aquatic Sciences 58:2271-2283.

Marshall, M., N. Piekielek, and J. DeCecco. 2004. Eastern Rivers and Mountains Network Park Summaries and Natural Resource Profiles. Appendix A to the ERMN Phase 1 Report.

May, C., R. Horner, J. Karr, B. Mar, and E. Welch. 1997. Effect of urbanization on small streams in the Puget Sound Lowland Ecoregion. Watershed Protection Techniques 2(4):483-494.

McAllister, L. S., B. E. Peniston, S. G. Leibowitz, B. Abbruzzese, and J. B. Hyman. 2000. A synoptic assessment for prioritizing wetland restoration efforts to optimize flood attenuation. Wetlands 20:70-83.

Minshall, G. W., K. W. Cummins, R. C. Petersen, C. E. Cushing, D. A. Bruns, J. R. Sedell, and R. L. Vannote. 1985. Developments in stream ecosystem theory. Canadian Journal of Fisheries and Aquatic Sciences. 42:1045-1055.

Mitsch, W. J., and J. G. Gosselink. 2000. Wetlands, 3rd edition. John Wiley & Sons, Inc., New York, NY.

Molloy, J. M. 1992. Diatom communities along stream longitudinal gradients. Freshwater Biology 28:59-69.

Montgomery DR. 1999. Process domains and the river continuum concept. Journal of the American Water Resources Association 35:397-410.

Morrison, M.L., B.G. Marcot, and R.W. Mannan. 1992. Wildlife-habitat relationships: Concepts and applications. University of Wisconsin Press. 343pp.

Mulholland, P. J., and E.J. Kuenzler. 1979. Organic carbon export from upland and forested wetland watersheds. Limnology and Oceanography 24:960-966.

Murphy, M. L. and J. D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat streams of the Cascade Mountains, Oregon. Canadian Journal of Fisheries and Aquatic Sciences 38:137-145.

NADP. 2003. National Acid Deposition Program 2003 annual summary. Http://nadp.sws.uiuc.edu/lib/data/2003as.pdf.

Natural Hazards Research and Applications Information Center. 1992. Floodplain management in the United States: An assesement report. TV-72105A, The Federal Interagency Floodplain Management Task Force, Boulder, CO.

Nelson, W. O. and P. G. C. Campbell. 1991. The effects of acidification on the geochemistry of Al, Cd, Pb, and Hg in freshwater environments: A literature review. Environmental Pollution 71:91-130.

Newbold, J. D., J. W. Elwood, R.V. O'Neill, and W. Van Winkle. 1982. Nutrient spiraling in streams: Implications for nutrient limitation and invertebrate activity. American Naturalist 120:628-652.

Niyogi, D. K., W. M. Lewis, Jr., and D. M. McKnight. 2001. Litter breakdown in mountain streams affected by mine drainage: Biotic mediation of abiotic controls. Ecological Applications 11:506-516.

Norokorpi, Y. E. A. 1997. Stand structure, dynamics, and diversity of virgin forests on northern peatlands. in C. C. Trettin, M. F. Jurgensen, D. F. Grigal, M. R. Gale, and J. K. Jeglum, editors. Northern Forested Wetlands: Ecology and Management. Lewis Publishers, New York, NY.

Novitzki, R. P. 1989. Wetland Hydrology. Pages 47-64 in S. K. Majumdar, R. P. Brooks, F. J. Brenner, and J. R. W. Tiner, editors. Wetlands Ecology and Conservation: Emphasis in Pennsylvania. The Pennsylvania Academy of Science, Easton, PA.

O'Connell, T. J., L. E. Jackson, R. P. Brooks. 2000. Bird guilds as indicators of ecological condition in the central Appalachians. Ecological Applications 10(6):1706-1721.

O'Connell, T. J., R. P. Brooks, S. E. Laubscher, R. S. Mulvihill, and T. L. Master. 2003. Using bioindicators to develop a calibrated index of regional ecological integrity for forested headwater ecosystems. Report No. 2003-01, Penn State Cooperative Wetlands Center, Final Report to U.S. Environ. Prot. Agency, STAR Grants Program, Washington, DC. 87pp.+app.

Oschwald, W. R. 1972. Sediment-water interactions. Journal of Environmental Quality 1:360-366.

Owen, H. J., and G. R. Wall. 1989. Floodplain management handbook. Contract No. WR18745467, U.S. Water Resources Council, Flood Loss Reduction Associates, Washington D.C.

Paul, M.J. and J.L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32: 333-365.

Patrick, W.H., Jr. and M.E. Tusneem. 1972. Nitrogen loss from flooded soil. Ecology 53:735-737.

Pennsylvania Environmental Council. 1973. Floodplain management: The prospects for Pennsylvania. Pennsylvania Environmental Council, Inc., Philadelphia, PA.

Peterjohn, W. T., and D. L. Correll. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. Ecology 65:1466-1475.

Phillips, J. D. 1989. Fluvial sediment storage in wetlands. Water Resources Bulletin 25:867-873.

Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime: a paradigm for river conservation and restoration. BioScience 47:769-784.

Pollino, C. A., P. Feehan, M. R. Grace, and B. T. Hart. 2004. Fish communities and habitat changes in the highly modified Goulburn catchment, Victoria, Australia. Marine and Freshwater Research 55:769-780.

Poole GC, 2002. Fluvial landscape ecology: addressing uniqueness within the river discontinuum. Freshwater Biology 47:641-660.

Prosser, D. J., and R. P. Brooks. 1998. A verified habitat suitability index for the Louisiana Waterthrush. J. Field Ornith. 69(2):288-298.

Radford, D. S. and R. Hartland-Rowe. 1971. A preliminary investigation of bottom fauna and invertebrate drift in an unregulated and a regulated stream in Alberta, Canada. Journal of Applied Ecology 8:883-903.

Reddy, K. R., G. A. O. Connor, E. Flaig, and P. M. Gale. 1999. Phosphorus retention in streams and wetlands: A review. Critical Reviews in Environmental Science and Technology 29:83-146.

Reice, S.R. 1984. the impact of disturbance frequency on the structure of a lotic riffle community. Verhandlungen der Internationalen Vereinigung fur Theoretische und Angewandte Limnologie 22:1906-1910.

Rheinhardt, R. D., M. C. Rheinhardt, M. M. Brinson, and K. E. F. Jr. 1999. Application of reference data for assessing and restoring headwater ecosystems. Restoration Ecology 7:241-251.

Richardson, C. J., and C. B. Craft. 1993. Effective phosphorus retention in wetlands: Fact or fiction? Pages 271-282 in G. A. Moshiri, editor. Constructed Wetlands for Water Quality Improvement. Lewis Publishers, Ann Arbor, Michigan.

Rocco, G. L., R. P. Brooks, and J. T. Hite. 2004. Stream plethodontid assemblage response (SPAR) index: development, application, and verification in the MAHA. Final Report. U.S. Environmental Protection Agency, STAR Grants Program, Washington, DC. Rep. No. 2004-01. Penn State Cooperative Wetlands Center, University Park, PA. 33pp+figs& app.

Rosemund, A. D., S. R. Reice, J. W. Elwood, and P. J. Mulholland. 1992. The effects of stream water acidity on benthic invertebrate communities in south-eastern United States. Freshwater Biology 27:193-209.

Rosgen DL. 1994. A classification of natural rivers. Catena 22:169-199.

Ross, R. M., R. M. Bennett, C. D. Snyder, J. A. Young, D. R. Smith, and D. P. Lemarie. 2003. Influence of eastern hemlock (*Tsuga Canadensis* L.) on fish community structure and function in headwater streams of the Delaware River basin. Ecology of Freshwater Fish 12:600-65.

Roth, Allan, and Erickson. Landscape influence on stream biological integrity assessment at multi-spatial scales. Landscape Ecology 47:141-156.

Sandin, L. and R. K. Johnson. 2000. The statistica power of selected indicator metrics using macroinvertebrates for assessing acidification and eutrophication of running waters. Hydrobiologia 422:233-243.

Schlesinger, W. H. 1997. Biogeochemistry: An Analysis of Global Change. Academic Press, New York, NY.

Schlesinger, W. H. and J. M. Melack. 1981. Transport of organic carbon in the worlds rivers. Tellus 33:172-187.

Schlosser, I. J. 1995a. Critical landscape attributes that influence fish population dynamics in headwater streams. Hydrobiologia 303:71-81.

Schlosser, I. J. 1995b. Dispersal, boundary processes, and trophic-level interactions in streams adjacent to beaver ponds. Ecology 76:908-925.

Schlosser, I. J. 1998. Fish recruitment, dispersal, and trophic interactions in a heterogeneous environment. Oecologia 113:260-268.

Scientific Assessment and Strategy Team. 1994. Science for floodplain management into the 21st century. Part V, Report of the Interagency Floodplain Management Review Committee to the Administration Floodplain Management Task Force, Washington D.C.

Scott, M. C., G. S. Helfman, M. E. McTammany, E. F. Benfield, and P. V. Bolstad. 2002. Multiscale influences on physical and chemical stream conditions across Blue Ridge landscapes. Journal of the American Water Resources Association 38:1379-1392.

Seitzinger, S. P. 1994. Linkages between organic matter mineralization and denitrification in eight riparian wetlands. Biogeochemistry 25:19-39.

Semlitsch, R. D. 2000. Principles for management of aquatic-breeding amphibians. Journal of Wildlife Management 64:615-631.

Semlitsch, R. D. and J. R. Bodie. DATE Are small, isolated wetlands expendable? Conservation Biology 5:1129-1133.

Shure, D. J., M. R. Gottschalk, and K. A. Parsons. 1986. Litter decomposition processes in a floodplain forest. The American Midland Naturalist 115:314-327.

Simberloff, D., I.M. Parker, and P.N. Windle. 2005. Introduced species policy, management, and future research needs. Front. Ecol. Environ. 3(1): 12-20.
Smith et al 1995

Smock, L. A. 1990. Spatial and temporal variation in organic matter storage in low-gradient, headwater streams. Archives of Hydrobiolgy 118:169-184.

Snodgrass, J. W., M. J. Komoroski, A. L. Bryan, Jr., and J. Burger. 2000. Relationships among isolated wetland size, hydroperiod, and amphibian species richness: Implications for wetland regulations. Conservation Biology 14:414-419.

Snyder, C. D., J. A. Young, and B. M. Stout, III. Landscape perspective on aquatic habitats in Canaan Valley, West Virginia. Northeastern Naturalist, (in review).

Snyder, C. D., J. A. Young, and R. Villella. 2003. Influences of upland and riparian land use patterns on stream biotic integrity. Landscape Ecology 18:647-664.

Snyder, C. D., J. A. Young, D. P. Lemarie, and D.R. Smith. 2002. Influences of eastern hemlock (*Tsuga Canadensis*) forests on aquatic invertebrate assemblages in headwater streams. Canadian Journal of Fisheries and Aquatic Sciences 59:262-275.

Snyder, C. D., J. J. Julian, and J. A. Young. 2005. Assessement of ambystomatid salamander populations and their breeding habitats in Delaware Water Gap National Recreation Area. Report to the National Park Service. Final Report submitted to the National Park Service, Delaware Water Gap National Recreation Area.

Soranno, P. A., S. L. Hubler, and S. R. Carpenter. 1996. Phosphorus loads to surface waters: A simple model to account for spatial pattern of land use. Ecological Applications 6:865-878.

Staubitz, W. W., and J. R. Sobashinski. 1983. Hydrology of Area 6: Eastern Coal Province, Maryland, West Virginia, and Pennsylvania. Open-file Report 83-33, U.S. Geological Survey, Water Resources Investigations, Towson, MD.

Strahler, A. N. 1952. Hypsometric (area-latitude) analysis of erosional topography. Bulletin of the Geological Society of America 63:1117-1142.

Stein, B. A., L. S Kutner, and J. S. Adams. 2000. Precious heritage. The status of biodiversity in the United States. Oxford University Press, NY. 399pp.

Taylor, J. R., M. A. Cardamone, and W. J. Mitsch. 1990. Pages 13-86 in J. G. Gosselink, L. C. Lee, and T. A. Muir, editors. Ecological Processes and Cumulative Impacts: Illustrated by Bottomland Hardwood Wetland Ecosystems. Lewis Publishers, MI.

Thorp, J. H., M. C. Thoms, and M. D. Delong. (in press) A model of biocomplexity in river networks across space and time. journal?

Tiner, R. W. 1988. Field guide to non-tidal wetland identification. Maryland Dep. Natural Resources, Annapolis, MD. 283pp.+app.

Tiner, R. W. 1998. In search of swampland. Rutgers University Press, New Brunswick, NJ.

Townsend CR. 1989. The patch dynamics concept of stream community ecology. Journal of the North American Benthological Society 8:36-50.

Townsend CR, Scarsbrook MR, Dolédec S. 1997. The intermediate disturbance hypothesis, refugia, and biodiversity in streams. Limnology and Oceanography 42:938-949.

U.S. Environmental Protection Agency. 2000. Mid-Atlantic Highlands Streams Assessment. EPA/903/R-00/015. U.D. Environmental Protection Agency Region 3. Philadelphia, PA. 64pp.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37: 130-137.

Vargo, S. M., R. K. Neely, and S. M. Kirkwood. 1998. Emergent plant decomposition and sedimentation: response to sediments varying in texture, phosphorus content and frequency of deposition. Environmental and Experimental Botany 40:43-58.

Verb, R. G. and M. L. Vis. 2000. Comparison of benthic diatom assemblages from streams draining abandoned and reclaimed coal mines and nonimpacted sites. Journal of the North American Benthological Society 19:274-288.

Vought, L. B.-M., J. Dahl, C. L. Pedersen, and J. O. Lacoursiere. 1994. Nutrient retention in riparian ecotones. Ambio 23:342-348.

Walker, L. R., and S. D. Smith. 1997. Community response to plant invasion. Pages 69-86 in J. O. Luken and J. W. Thieret, editors. Assessment and Management of Plant Invasions. Springer, New York, NY.

Wallace, J. B., J. R. Webster, S. L. Eggert, J. L. Meyer, and E. R. Siler. 2001. Large woody debris in a headwater stream: long-term legacies of forest disturbance. International Review of Hydrobiology 86:501-513.

Wallace, J. B., J. R. Webster, and J. L. Meyer. 1995. Influence of log additions on physical and biotic characteristics of a mountain stream. Canadian Journal of Fisheries and Aquatic Sciences 52:2120-2137.

Wallace, J. B., T. F. cuffney, C. C. Lay, and D. Vogel. 1987. Influence of an ecosystem-level manipulation on prey consumption by a lotic dragonfly. Canadian Journal of Zoology 65:35-40.

Ward JV. 1989. The four-dimensional nature of lotic ecosystems. Journal of the North Atlantic Benthological Society 8:2-8.

Ward JV, Stanford JA. 1983. The intermediate disturbance hypothesis: An explanation for biotic diversity patterns in lotic ecosystems. Pages 347-356 in: Fontaine TD, Bartell SM (eds.), Dynamics of lotic ecosystems, Ann Arbor Science Publishers, Ann Arbor, MI, USA.

Ward JV, Tockner K, Arscott DB, Claret C. 2002. Riverine landscape diversity. Freshwater Biology 46:807-819.

Ward JV, Tockner K, Arscott DB, Claret C. 2002. Riverine landscape diversity. Freshwater Biology 47:57-539.

Wardrop, D. H., and R. P. Brooks. 1998. The occurrence and impact of sedimentation in central Pennsylvania wetlands. Environmental Monitoring and Assessment 51:119-130.

Wedin, D. A., and D. Tilman. 1990. Species effects on nitrogen cycling: A test with perennial grasses. Oecologia 84:433-441.

Wiens JA. 2002. Riverine landscapes:taking landscape ecology in the water. Freshwater Biology 47:501-515.

Williams, D. J., D. Bland, M. C. Williams, R. W. Patrzalek, L. Banks, B. Logue, and M. T. Price. 1994. Role of wetlands in water quality maintenance: Microcosm investigation of factors influencing the buffering capacity of urban riparian wetlands. Lake and Resevoir Management 9:124.

Woods, K. D. 1997. Community response to plant invasion. Pages 56-68 *in* J. O. Luken and J. W. Thieret, editors. Assessment and Management of Plant Invasions. Springer, New York, NY.